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Planning and decision-making in human-wildlife conflict and coexistence

Edited by
Silvio Marchini, Alexandra Zimmermann, Katia Maria P. M. B. Ferraz and Jenny Anne Glikman

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Planning and decision-making in human-wildlife conflict and coexistence

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Mapping Micro-Level Decision-Making for Alligator Management in North Carolina

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Human-carnivore interactions represent a grand challenge to conservation decision-making and legitimacy across all levels of governance. Human populations continue to encroach upon and devastate carnivore habitats and populations, intensifying interactions between a variety of biodiversity interests and beneficiaries. As a result, carnivores most intensely impact those living in their midst, demanding increased attention by local decision makers, who are often best suited to catering to the needs of communities most affected. Their views and desires can serve as a forerunner of public trust and acceptance of policies created. However, due to the complexity of decisions about carnivores, these actors are often overlooked in the formal decision process. To address this need, we applied multi-criteria decision analysis (MCDA) to a case study of American alligator (*Alligator mississippiensis*) conservation in 10 coastal North Carolina counties to identify and postulate legitimate outcomes. We surveyed 25 local decision makers who are or may be responsible for management decisions concerning the American alligator and asked them to evaluate and indicate the level of importance of salient alligator management elements. Results indicate that decision makers strongly favored the wildlife and social factors when making alligator management decisions, as well as the criteria human well-being, attitudes toward alligators, education programs, and storm mitigation. Respondents favored highly managed and balanced management alternatives to maximize preferred criteria and achieve legitimate alligator management at the local level. These results demonstrate that local decision makers are capable of identifying what is important to alligator management decisions, and can provide an insightful look at trade-offs that need to or could be made to achieve optimal alligator outcomes. We conclude that local decision makers should become more involved in shaping carnivore outcomes to enhance legitimacy of alligator policy and help achieve conservation targets. Future research will need to further expand understandings of local decision makers' decision-making process in other carnivore contexts. Researchers will want to consider using and refining decision analysis to cut through the complexity of carnivore conservation decision-making that exists across wide geopolitical expanses.

Keywords: alligator, analytic hierarchy process, carnivore, coexistence, decision-making, multi-criteria decision analysis

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INTRODUCTION

Human populations continue to encroach upon and devastate habitats, intensifying interactions between a variety of biodiversity interests and beneficiaries. These trends demand increased attention by local decision makers, who often have an important role in conservation outcomes (e.g., Press et al., 1996, p. 1547; Doyle-Capitman et al., 2018, p. 376). These actors are often best suited to catering to the needs of communities most affected by human-wildlife interactions (Devas and Grant, 2003, p. 307). Hence, their views and desires can serve as a forerunner of public trust and acceptance of policies created (Sjölander-Lindqvist et al., 2015, p. 179) as well as dissent (Redpath et al., 2017, p. 2161). Although plurality of voice is a cornerstone of democratic biodiversity conservation governance, the possibility exists that divergent or overlooked actors and their views, and as well as shifting social and ecological realities, may prevent outcomes that are optimal for both human and non-human species.

Human-carnivore interactions represent a grand challenge to conservation decision-making and legitimacy (Primm and Clark, 1996, p. 1037; Messmer, 2000, p. 1000; Serenari and Taub, 2019, p. 1). The challenge is formidable as positive and negative interactions can strike the core of societies – rousing a range of epistemologies, social and ecological values, perspectives, problem definitions, and solutions (Dickman, 2010, p. 463). Carnivores most intensely impact those humans living in their midst (Serenari et al., 2018, p. 363). For instance, carnivore species worldwide are known for causing economic loss by preying on livestock (Treves and Karanth, 2003, p. 1492) and can pose a significant threat to human safety through vehicle collisions, disease, or direct attacks on humans (Riley and Decker, 2000, p. 51). Therefore, it is arguably intuitive that governing entities should consider the views of those most impacted by interactions with carnivores, as they are often most effective at contributing to and formulating viable decisions and outcomes (Devas and Grant, 2003, p. 306; van der Ploeg and van Weerd, 2004, p. 346; Sjölander-Lindqvist et al., 2015, p. 180).

However, local stakeholders rarely have a say in carnivore conservation outcomes because these species are often held in the public trust, and decisions are made by state and federal authorities (e.g., wildlife or animal health agencies) (Redpath et al., 2017, p. 2158). This omission is a critical oversight in the instance of American alligators (*Alligator mississippiensis*). For instance, the designation of an alligator as a nuisance is often left up to subjective judgments about an animal's behavior or level of danger it poses to humans' livelihood (Johnson et al., 1985, p. 96; Hayman et al., 2014, p. 489), and decisions to leave or remove it from a pond governed by a homeowner's association, golf course, or local park are often left to the space's managing institution; these decisions may be guided by legal liability (Connaughton et al., 2002, p. 74), public response to alligator presence (Jacobsen and Kushlan, 1986, p. 188), or overt threats to public safety (Eversole et al., 2014, p. 15). We note that inclusion of formal and informal local decision makers and how they negotiate carnivore interactions complicates existing carnivore governance arrangements. Hence, picking the appropriate decision-making tool is critical to simplify the complexity of challenges presented by diverse decision-making scenarios and achieve ideal management outcomes (Bower et al., 2018, p. 2).

Given that carnivore management is notoriously contentious and multifaceted, tools that evaluate the tradeoffs and co-benefits of different management actions help reveal optimal carnivore management solutions and enhance the legitimacy of those decisions (Lundmark and Matti, 2015, p. 150; Robinson et al., 2016, p. 2). In short, the success of human-carnivore cohabitation schemes and related management relies on the ability to integrate the array of salient social and ecological factors that influence the decision-making process. Multi-criteria decision analysis (MCDA) is one promising method that can help navigate the complexities of decision-making and secure optimal carnivore conservation outcomes (Adem Esmail and Geneletti, 2018, p. 43). It has demonstrated success in identifying potential solutions to natural resource management problems (Redpath et al., 2004, p. 357; Driscoll et al., 2016, p. 202). Despite its utility to elicit socio-ecological tradeoffs in other environmental contexts, MCDA has been underutilized to help resolve conflicts in human-carnivore decision-making (Redpath et al., 2004, p. 351).

We address this need by applying MCDA in a carnivore management context to investigate the decision-making process at the local level in North Carolina. The case of the alligator in eastern North Carolina is a useful place to begin developing systematic understandings of local-level decision-making about carnivores because interactions between people and alligators have become a socially and politically contentious carnivore management issue at the county and municipality levels. Moreover, policy conflict is often a prerequisite for decision analysis in wildlife management (Redpath et al., 2004, p. 358; Riley and Gregory, 2012, p. 103). As we aim to demonstrate, local decision makers are often involved in resolving or stoking such conflict and shaping outcomes.

METHODS

Study Area

The American alligator ranges from the southern tip of Texas along the Gulf Coast to Florida and continues northward along the Atlantic Coast to North Carolina. Historically, alligator populations located in North Carolina are considerably smaller than their southern counterparts. This is due to the cooler annual climate and lower temperatures of North Carolina, which cause slower maturation and, consequently, lower reproduction rates (Gardner et al., 2016, p. 545). Rising sea levels and human expansion have depleted alligator habitat (Carle, 2011, p. 1276; Gardner et al., 2016, p. 541), causing alligators to move further inland in search of food, shelter, and mates and thus interact with humans more frequently (Eversole et al., 2014, p. 15).

Alligators are a state trust resource and are managed by the North Carolina Wildlife Resources Commission (NCWRC, 2017, p. 11). This study focused on local-level formal and informal decision makers within the 10 coastal counties of North Carolina's Alligator Management Unit 1 (AMU 1): Brunswick, Carteret, Columbus, Craven, Hyde, Jones, New

Hanover, Onslow, Pamlico, and Pender. According to the NCWRC's North Carolina Alligator Management Plan (AMP), AMU 1 comprises counties within North Carolina that harbor the most robust alligator populations and quality alligator habitat [e.g., rivers, lake, and estuaries [Gardner et al., 2016]]. As a result, human-alligator interactions are more frequent in these areas (NCWRC, 2017, p. 9). The North Carolina AMP proposed a list of parties that could potentially be impacted by alligator management strategies (NCWRC, 2017, p. 24). Using this list we categorized formal decision makers as those with local governmental positions and informal decision makers as individuals within nongovernmental organizations such as public or private businesses or groups.

The NCWRC is aware of the role that local communities play in helping maintain the viability of the country's northernmost alligator population, which is smaller and more easily perturbed than southern populations. The agency often works closely with local officials to communicate about alligator management, and may also help a local community achieve its alligator management goals, such as targeted removal of nuisance alligators or improving public knowledge of living with alligators (NCWRC, 2017, p. 8). Correspondingly, there is a need to minimize any threats posed by alligators to the public or to alligators by the public. Policy conflict over alligator management came to a head in 2018 when the NCWRC approved permits to hunt alligators in AMU 1 (NCWRC, 2017, p. 32). The decision was hailed by hunting and public safety proponents but rebuked by segments of the public and some local officials[1,2].

Study Design

Our quantitative survey focused on AMU 1 formal [governmental (e.g., county, city council, police)] and informal (e.g., golf course general manager, HOA board member) decision makers. Novel decision-making tools that objectively evaluate the range of interests and potential outcomes can inform policy conflict resolutions and do so with small sample sizes (Robinson et al., 2016, p. 2; Darko et al., 2019, p. 447). We employed the Analytic Hierarchy Process (AHP), a MCDA method, because it has the ability to quantify the priorities of decision makers through comparisons of explicit criteria and management alternatives in a manner that is replicable and transparent (Adem Esmail and Geneletti, 2018, p. 43) and provide useful insights when samples are small (Darko et al., 2019, p. 447). We focused study questions on eliciting salient elements of decision-making (i.e., factors, issues, and criteria) concerning alligator management.

Decision Tree and Survey Instrument Development

We began by interviewing local-level decision makers who had previously been responsible for making decisions about alligator outcomes within their jurisdiction to elicit which elements decision makers judged as most valuable. Recruitment for interviews occurred by first creating a master list of potential contacts via a Google™ search using a combination of the search terms *alligator, sightings, hunting,* and *conflict* to identify any decision makers publicly mentioned in past human-alligator interactions or policy decisions in the target counties ($N = 33$). We contacted potential participants by alternating between email and phone calls weekly during August and September, 2019 (Dillman et al., 2014, p. 285), and carried out all subsequent interviews by phone. We first asked informants to elaborate on their occupation and the role they played in making decisions regarding alligator management to ensure that respondents were responsible for alligator management decisions as a part of their occupation. We then asked them to rank and discuss five primary *factors* in terms of importance when making decisions regarding alligator management: social, political, economic, alligator-specific, and landscape-specific, as well as each factor's corresponding *issues* and *criteria*. We conducted four interviews due to constraints caused by Hurricane Dorian in September 2019. The four interview participants consisted of a golf course manager, county commissioner, chief of police, and county manager; three of which had previously made decisions regarding alligators as a part of their job, while the fourth indicated that they would be responsible for alligator management decisions.

We supplemented interviews with a thorough literature review. We searched the Google Scholar database, for relevant literature using a combination of the following search terms: *wildlife, management, carnivore, alligator, decision-making, factors,* and *element/factor*. We considered only articles that directly related to the topic of carnivore management and specifically discussed factors that influence decision-making. We focused our review on peer-reviewed ($n = 44$) and gray literature (e.g., books and reports, $n = 5$), and thematically coded (Guest et al., 2011) to elicit primary decision-making elements. We found that researchers largely focused on topics such as public risk assessment and attitudes toward carnivores (e.g., Riley and Decker, 2000, p. 58; Smithem and Mazzotti, 2008, p. 10), providing education about carnivores (Eversole et al., 2014, p. 19; Skupien et al., 2016, p. 274), human development and associated impact on human-carnivore interactions (Patterson et al., 2003, p. 172; Eversole et al., 2018, p. 7), and the role of wildlife systems in decision-making (Liu et al., 2016, p. 21; Expósito-Granados et al., 2019, p. 9) at the local level. Employing a tripartite coupled human and natural systems framework (Liu et al.'s, 2016, p. 16), the first level of the decision tree consisted of Social, Natural, and Wildlife factors. We renamed landscape- and alligator-specific factors Natural and Wildlife, respectively, to be more inclusive of the issues and criteria mentioned in the interviews. The literature maintained the diminished importance of economic and political factors in local-level alligator management decisions and were, thus, excluded from the decision tree design. The second branch of the decision tree consisted of issues specific to each factor, and the third branch consisted of criteria belonging to each issue (**Figure 1**). The final version of the decision tree aligned with the goal of enhancing the legitimacy of alligator management decision-making at the local level, understanding that legitimate decision-making promotes the inclusion of relevant stakeholders and experts, is transparent and reliable, and produces quality management practices (Serenari and Taub, 2019, p. 2).

[1]https://www.newsobserver.com/news/state/article216431885.html
[2]https://www.starnewsonline.com/news/20180402/nc-towns-yet-to-take-bait-on-alligator-hunting

We created an online survey in Qualtrics (2020). The survey consisted of pairwise comparisons at each branch of the decision tree to determine the relative importance of each element of alligator management decision-making. To indicate their preference for considering elements when making decisions, we used the Saaty scale (1 = equally preferred, 9 = extremely preferred, Saaty, 2008, p. 86). Using this scoring system, the participants then also compared factors, issues, and criteria among each branch in the tree, and then again between one branch and the elements corresponding directly below it. We then provided decision makers with three management options (alternatives) and asked them to indicate their preference of management to maximize, or enhance, each individual criterion of the decision tree.

Alternatives represented a range of management intensities, which is common logic used in wildlife management: highly managed, balanced, and land-sharing (Redpath et al., 2004, p. 354). Highly managed practices often indicate policies that maximize human benefits (e.g., safety, development) and may be considered equivalent to a zero-tolerance policy of alligators inhabiting space near human settlements. Lethal control or removal is preferred and carried out by governing agencies. Hence, our alternatives characterized a sliding scale of aggressiveness employed by carnivore managers (Serenari, 2020), specifically, alligator management based on frequency of sightings, risk perception, proximity to populated areas, alligator density, predatory behavior of alligators, and governance arrangement (**Table 1**) (e.g., Johnson et al., 1985, p. 100; NCWRC, 2017, p. 7). A balanced alternative represented equal consideration of human and alligator needs (e.g., prey, habitat). As an example, AMU 1's permitted alligator hunt, in which municipalities can collaborate with the NCWRC to determine alligator population sizes, places of concern for public safety, areas in which alligator hunts can be conducted safely, and number of permits to issue. Land-sharing signifies management practices that largely consider the needs of alligator populations, in which the suggested form of management is to "leave them be," regardless of where the alligators are located. Individuals who report alligators to the NCWRC are provided information regarding alligators and their behavior and advised to leave the alligator(s) alone until it moves from its current location (NCWRC, 2017, p. 7). We provided definitions of each element in the decision tree, as well as the different outcomes of each alternative to respondents to ensure consistent interpretation of meanings (see **Supplementary Materials**).

Recruitment and Survey Administration

We conducted two phases of recruitment of formal and informal local-level decision-makers by email and phone. Formal decision makers were located in AMU 1 counties and included city or county governments, police departments and informal included HOA board members, golf clubs, park rangers, and conservation organizations. The first phase of recruitment occurred every 3–4 weeks between August and December 2019, and the second

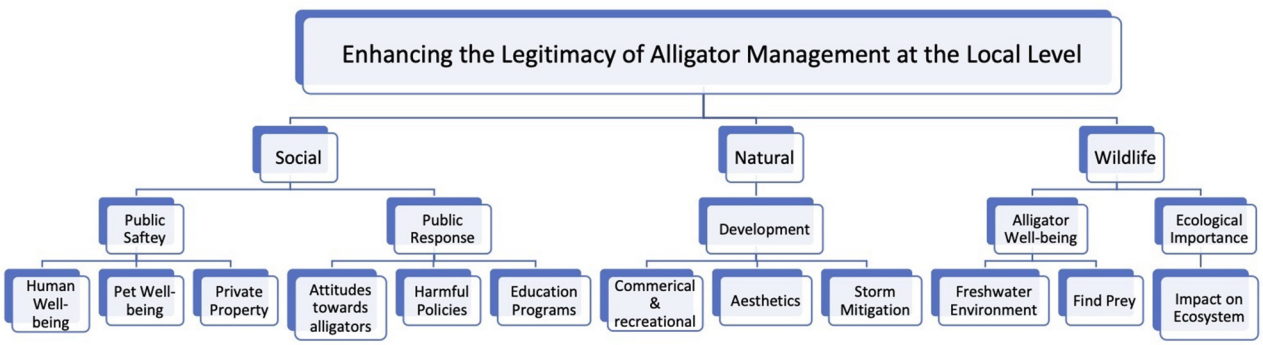


FIGURE 1 | Hierarchal decision tree consisting of the goal (top branch), three factors (second branch), five issues (third branch), and 12 criteria (bottom branch).

TABLE 1 | Management options (alternatives) of alligator management.

Option	Description	Frequency of sighting	Risk perception	Proximity to populated areas	Density	Predatory behavior (of alligators)	Governance
A	Highly Managed	Decreased	Minimal	<25 mi from shore to human activity	Low (0–2/km to water's edge)	Limited	Centralized (state as main governing body)
B	Balanced	Random	Limited	Some human activity	Medium (3–5/km to water's edge)	Regulated	Public-Private [co-governing between state and stakeholders (biologists)]
C	Land-Sharing	Increased	High	Anywhere	High (≥6/km to water edge)	Unrestrained	Interactive (collaborative governing between state, stakeholders, and citizens)

Options from A–C represent declining levels of management intensity & the resulting effects of its respective management style.

phase increased contact frequency, occurring biweekly between March and July 2020 (Dillman et al., 2014, p. 336). We terminated recruitment efforts using a 7-attempt callback/email design (Dillman et al., 2014, p. 285). We expanded recruitment efforts to include decision makers that had previously been involved in alligator management decision-making, as well as those that would be responsible for these decisions in the future. We screened participants using the following questions: "As part of your job, do you have previous experience making decisions about alligators (e.g., decision to relocate, leave alone)?" and "As part of your job, is it possible you may someday make a decision about an alligator (e.g., decision to relocate, leave alone)?". We invited decision makers who answered *yes* to the first question and *yes* or *maybe* to the second question to further participate in the survey. If they responded *no* to both questions, they were excluded from the study. We also employed chain referral sampling (Etikan et al., 2015, p. 1) to increase our chances of reaching the person within each organization responsible for alligator management decisions (see **Supplementary Materials** for complete timeline of interview and survey recruitment).

The study (#6291) was approved by the Texas State University Institutional Review Board on April 8, 2019.

Analysis

Using the Saaty scale, scores were attributed to each element based on the preference indicated by the respondent and placed into a matrix. The diagonal values within the matrix equal 1.0 as they represent comparisons of the same element (Harputlugil, 2018, p. 224). Following (Saaty, 2008, p. 85), the element with the greatest assessed importance was assigned the score provided by the respondent, and the reciprocal of that score was given to the corresponding comparison. We first calculated the sum of each column, and then divided each given value by the sum of its respective column to determine the priority weights of each element. Then, we summed calculated values across each row and divided by the total number of elements within each matrix to create the priority weight of each element. This process occurred for each individual respondent's survey responses. Following the calculation of the priority weights as determined by each individual respondent, the average of the weights for each element was taken. When added, the weights of all elements that belong to the same parent element directly above them equals 1.0.

In addition to the average weights attributed to the three management alternatives with respect to each criterion, the global weights of the alternatives were also calculated to determine their relative importance with respect to the overall goal. First, the global criterion weights were calculated by multiplying the averaged individual weight of each criterion by the weights of its respective issue and factor. Next, the global criterion weights were multiplied by the averaged individual alternative weights with respect to each criterion. Lastly, the total of the calculated weights for each alternative was taken to create the final global priority of the three alternatives.

We did not conduct any official collection of nonresponse data because potential respondents became unresponsive. Of the 97 individuals that opened the survey, 72 either responded *no* to both screening questions or did not complete the survey. Moreover, some potential respondents asked us to not contact them again because they were not involved in alligator management decisions, did not have any alligators within their jurisdiction, or were unavailable due to the pandemic. Thus, we posit that most potential respondents that we contacted did not or would not make such decisions.

RESULTS

A total of 97 individuals invited to participate completed at least part of the survey, and 25 provided valid MCDA results for analysis (response rate of 30%). The majority of respondents were 55 years or older, lived in suburban areas, and had a bachelor's degree or higher. More than half of the survey respondents were employed in city or town government, with the remaining participants employed in county government, law enforcement, property management, and others (**Table 2**). Of the 25 respondents, 14 indicated previous experience in making decisions about alligators as a part of their job, while the remaining respondents indicated they would be responsible for any decisions regarding alligator management in the future.

When making decisions regarding alligator management, respondents indicated that the wildlife factor (weight = 0.38) was most important for consideration, followed by social (0.37), and natural (0.25). Within the wildlife factor, about half of the respondents (i.e., 13/25) claimed that alligator well-being and ecological importance of alligators were of equal importance in making a decision regarding alligator management. This finding was reflected in the averaged score of the two issues at 0.55 and 0.45, respectively. Similarly, 20 out of the 25 respondents claimed no difference between the wildlife criteria freshwater environment (0.55) and find prey (0.45). The wildlife criterion impact on ecosystem was the sole criterion of ecological importance and did not undergo a pairwise comparison, and thus resulted in a weight of 1.0.

Concerning the social and natural factors, respondents strongly rated public safety (0.72) over public response (0.28), with seven respondents claiming equal importance of the two issues in decision-making. Only two participants answered that public response was more important. Respondents prioritized the social criteria human well-being (via public safety) (0.59), followed by education programs (via public response) (0.51), and attitudes toward alligators (via public response) (0.34).

TABLE 2 | Occupational makeup of survey participants.

Occupation	*N* =
City/Town Government	15
County Government	3
Law Enforcement	2
Property Manager (HOA, golf course, campground)	2
Park Ranger	1
Conservancy	1
Total	**25**

Storm mitigation was the highest rated natural criterion (0.61) (**Figure 2**).

Respondents indicated that highly managed was the preferred management method for maximizing social criteria (private property, education programs) and natural criteria (aesthetics, storm mitigation). Respondents preferred a balanced management approach, maximizing all wildlife-specific criteria, most of the social criteria, as well as the natural criterion commercial and recreational development (**Table 3**).

Balanced management (0.37) was the most preferred alternative to achieving the overall goal of enhancing the legitimacy of alligator management decision-making at the local level, closely followed by highly managed (0.35). Land-sharing was ranked last to maximize all criteria except for human well-being and attitudes toward alligators. In both cases, land-sharing was greater than highly managed by <0.01.

DISCUSSION

Respondents appear attuned to the role alligators play in local social and ecological systems. Decision makers in our study demonstrated awareness of and agreed on the importance of the wildlife system in the carnivore management decision-making process. Though slightly more important to decision-making than other factors, respondents offered a balanced view, giving nearly equal weight to alligator welfare and the role of alligators in local ecosystems. There are three distinct explanations for why formal and informal local decision makers held the alligator system in high regard. First, wildlife can be an integral part of the identity of some regions and localities (Treves et al., 2006, p. 387). For instance, residents in Florida favor the presence of alligators, as it is seen as an indicator of a healthy environment (Smithem and Mazzotti, 2008, p. 15). In this case, eastern North Carolina is known for its wild landscapes (Serenari et al., 2018, p. 361), and it is conceivable that respondents' answers reflect this internalization. Second, local decision makers may be considering how their communities need to adapt to novel social and ecological dynamics to increase the legitimacy, communication, and understanding of their decisions among the public (Patterson et al., 2003, p. 173; Lundmark and Matti, 2015, p. 147). Dynamic change is occurring to North Carolina's coasts. For instance, human activities, sea level rise, and saltwater intrusion are having and expected to have substantial impacts on social-ecological systems, particularly wildlife distribution and abundance and human-wildlife interactions (Bhattachan et al., 2018, p. 127). Considerations for these realities and growing emphasis on legitimacy, salience, and empowerment of local decision-making may have influenced our findings (also evidenced by preference for storm mitigation qualities of alligator habitat). Finally, public cognitions concerning large carnivores can play an important role in the decision-making process (Lute and Attari, 2017, p. 139). Researchers have theorized that values oriented toward wildlife are shifting from the sole belief that wildlife are to be used for human benefit to the idea that wildlife are meant to be appreciated and respected, particularly among urbanizing human populations (Manfredo

TABLE 3 | Average of individual AHP weights for alternatives related to each criterion.

Criterion	Highly managed	Balanced management	Land sharing
Human Well-being	0.29	0.42	0.30
Pet Well-being	0.34	0.37	0.29
Private Property	0.37	0.36	0.27
Attitudes toward Alligators	0.27	0.43	0.30
Harmful Policies	0.35	0.36	0.29
Education Programs	0.36	0.34	0.30
Commercial & Recreational Development	0.37	0.38	0.25
Aesthetics	0.38	0.35	0.27
Storm Mitigation	0.47	0.35	0.22
Freshwater Environment	0.35	0.38	0.27
Find Prey	0.31	0.39	0.30
Impact on Ecosystem	0.33	0.35	0.32

Balanced management was most preferred among participants for maximizing eight of the 12 criteria, and highly managed was preferred for the remaining four criteria.

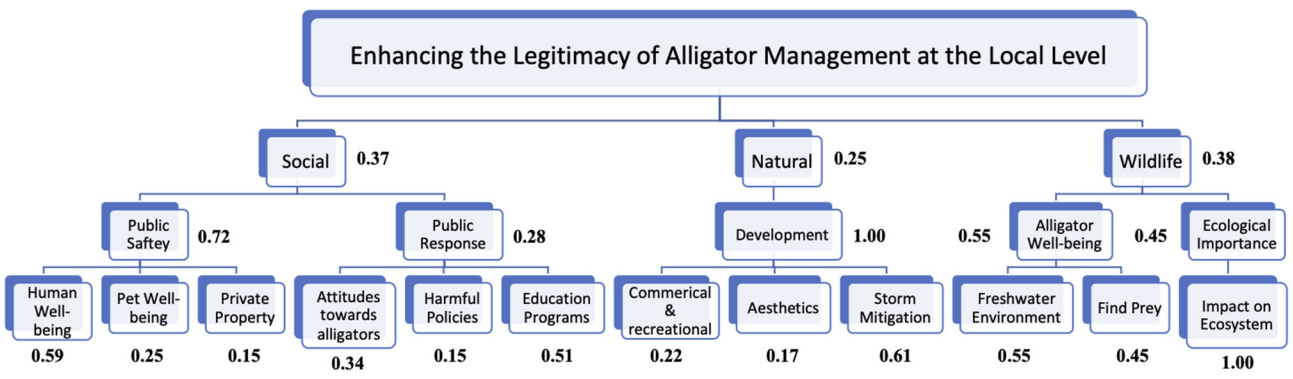


FIGURE 2 | The hierarchal decision tree contains the final AHP weights of all factors, issues, and criteria, indicating which factor, issues, and criteria were most preferred by respondents. Final weights were calculated by taking the average of all individual respondent weights.

et al., 2020, p. 7). However, we note that this research occurred on the heels of public outcry over the NCWRC's proposed hunting of alligators in 2018; therefore, our results may reflect an underlying preference for coastal communities to share space with alligators, but with caveats. While public outcry may have roused strong policy preferences among respondents, responses were anonymous. Therefore, social desirability bias should be of little concern in this study, but future research will want to consider how public outcry over carnivore policies might impact local-level decision-making.

Human welfare was a top priority for local-level decision makers in our study, specifically, human well-being linked to public safety (e.g., health, safety, and social relations). Previous studies demonstrate that concern for public safety is a top factor in decision-making at all levels of governance and is underpinned by risk perceptions (Riley and Decker, 2000, p. 58; Gore et al., 2006, p. 40; Smithem and Mazzotti, 2008, p. 19). Although perceived risks concerning alligators may be attributed to social and cultural beliefs (Dickman, 2010, p. 459) or situational factors such as living near water or having children or pets (Hayman et al., 2014, p. 484), many times they evolve from exaggerated ideas about alligator behavior (Eversole et al., 2014, p. 17). This paradoxical nature of perceived risk may throw a wrench into plans to integrate alligators into the coastal North Carolina landscape, particularly if policies address atypical alligator behavior (Rogers, 2011, p. 293). The status of alligators on the North Carolina coast is at a critical juncture, and intentional educational opportunities that create "Alligator-Wise"[3] initiatives at the community and regional scales would be invaluable to help communities promote novel perceptions of risk grounded in empirical data and probability.

Respondents also considered public response in their decisions, but to a small degree. Similar to the NCWRC, the decision makers within our study placed great importance on public education about alligators. One goal of the NCWRC AMP is to provide comprehensive knowledge of alligators and their management to the public through education and outreach strategies. Some proposed strategies include formal and informal public forums, technical guidance to landowners and managers, and educational information dispersed through various media outlets (NCWRC, 2017, p. 8). Research demonstrates that deficient knowledge of carnivores and their ecological importance can be detrimental to promoting coexistence between humans and carnivores (Lute and Gore, 2014, p. 1065; Expósito-Granados et al., 2019, p. 9). Future consideration should be given to finding ways to involve local decision makers in designing and administering educational opportunities to promote human-alligator coexistence. Our study suggests that content should include integrating net positives that benefit humans, alligators, and the coastal ecosystem such as protecting habitat which would in turn buffer communities from increasingly intense storms (Gedan et al., 2011, p. 8).

If local decision makers are to become more involved in carnivore management outcomes, as we argue they should, we must better understand the ideology underpinning decisions and examine how they align with alternatives in use to provide a clearer picture of the efficacy of those alternatives, as well as full consideration for novel alternatives and arrangements where necessary. Local decision makers in our study favored a landscape with a strong human presence to moderate interactions between humans and alligators and to maximize human activity and protect private property from damage. Yet, a balanced management plan was preferred to maximize human well-being, human attitudes, and the remaining suite of criteria. These results speak to the heart of the so-called predator paradox (Shivik, 2014), empowered by the mass media and characterized by a persistence of memory and perpetuation of an unwillingness to alter our imaginations about how to live peacefully with carnivores (Debord, 1967). Retaliation, separation, and aggressive carnivore management are often the rallying cry and default outcome when human safety and damage to private property are a major concern (Treves and Karanth, 2003, p. 1492; Lute and Attari, 2017, p. 140). Critics argue that the ideas about how to live with carnivores on an increasingly crowded planet require upgrading (Carter and Linnell, 2016, p. 577; López-Bao et al., 2017, p. 1; Lute et al., 2018, p. 231; Serenari, 2020, p. 7), as the system of ideas that embrace and promote total separation of humans and carnivores are deficient in a holistic understanding and appreciation of carnivores (Skupien et al., 2016, p. 266). Carnivore populations are decreasing as a result of carnivores leaving their natural habitat range due to loss of habitat, as seen in the alligator populations in North Carolina. Consequently, there is a growing necessity to incorporate decision-making tools that allow for impartial and equal consideration of all relevant criteria to create legitimate alligator management. Our study suggests that local-level decision makers may provide an insightful look at trade-offs that need or could be made to achieve optimal alligator and other carnivore conservation outcomes, promoting local understanding and support, and ultimately legitimacy, of the decision-making process (Doyle-Capitman et al., 2018, p. 379).

Use and Limitations of MCDA

The use of decision modeling in this study allowed quantification of the decision-making process of alligator management at the local level. Unlike attitudinal measures, participants were able to analyze and compare multiple elements at one time. Though the method has great potential, administering the study's scope and MCDA across a broad scale requires refinement. We hope to elaborate on these lessons in a future paper, but provide a brief overview here. First, due to the exploratory nature of the study, a roadmap for preparing the sampling frame did not exist. Hence, overall participation was voluntary and based on a convenient sample. Additionally, recruitment for this study was hampered by forces out of our control, including Hurricane Dorian (2019), Tropical Storm Arthur (2020), and the COVID-19 pandemic (2020). Second, we achieved greater participation from government officials than from informal decision makers. Therefore, our results may reflect biases in this regard. Third, though representativeness was not the goal of this study, only one-fourth of respondents who began the survey completed the

[3]https://www.santafenewmexican.com/news/we-saved-the-alligators-then-moved-onto-their-turf/article_a4defe7d-0d70-58b9-9fd8-8ec7b56f743b.html.

MCDA portion (although it was placed in the first third of the survey). We attribute this outcome to the cumbersome nature of the MCDA design and applying it across a wide geospatial area rather than in a collective setting (e.g., workshop). Fourth, the broad scale of our research may have resulted in survey question design that lacked context or underrepresented the suite of factors, issues, and criteria specific to the local scale in North Carolina. For example, during the interview process, informants revealed that economics and politics received little consideration when making decisions about alligator management, and thus these factors were not included in our survey. These factors were claimed to be irrelevant to decision makers in North Carolina's AMU 1 as well as the broad alligator literature. Nevertheless, future studies should contextualize factors to the best of their ability and strive for representativeness. Finally, although there was sufficient data to obtain useful insights into local-level decision makers' preferences for alligator management using AHP, the small sample size did not allow for comparisons among different groups of formal and informal decision makers. Out of the 25 respondents, 20 were formal decision makers and five were informal decision makers. The small sample sizes of formal and informal decision makers separately prevented us from making any assumptions on the individual groups that would accurately represent the individual groups. Also, due to the imbalance between the two group sizes, we were unable to compare formal and informal decision-making processes for alligator management. Future research would benefit from a concerted effort to recruit formal and informal decision makers to achieve a richer representation of their decision process, as well as further exploration in comparisons between the decision-making process of formal and informal decision makers.

CONCLUSION

The purpose of our study was to highlight the current gap in carnivore management decision-making, specifically at the local level. We applied MCDA to give voice to those decision-makers who are often overlooked in the carnivore management decision-making process. We surveyed formal and informal decision makers in eastern North Carolina and asked them to equally consider principle elements of alligator management. Our study highlights that these critical actors considered the wildlife and social factors when making decisions, and preferred either balanced or highly managed practices to manage for alligators. While carnivore management policies created at the macro-level are essential to carnivore conservation, coexistence between humans and carnivores is best promoted when these large-scale policies can be adapted to specifically target the needs of the local people. Understanding what local-level decision makers consider important when managing carnivores is critical to increasing local involvement in the decision process and ultimately improving the legitimacy of management policies.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by Texas State University Institutional Review Board (#6291). The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

RC and CS conceived and developed the project and collected data. RC, CS, and DK analyzed the data and wrote and edited the manuscript. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.650339/full#supplementary-material

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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When a Shark Is More Than a Shark: A Sociopolitical Problem-Solving Approach to Fisher-Shark Interactions

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Fisheries are often conceptualized through a biophysical lens resulting in management approaches that fail to account for stakeholder conflicts and sociopolitical inequities. Using a fisher engagement approach, this case study examines the sociopolitical dimensions of fisher-shark interactions in pursuit of more complete problem definitions and effective solutions. Through interviews with Hawai'i small boat fishers and observations of a community-based shark-tagging project, we examined fisher perspective, socioeconomic landscapes, stakeholder relationships, and power dynamics. We interpreted these data using an adapted framework that mobilizes concepts from conflict theory and problem definition. We discovered that economic cost, sharks as fishing competitors, and factors of fishers' on-the-water decisions define the fisher-shark interaction problem at the dispute level. Deeper conflicts include fishers' poor perceptions of management legitimacy, degraded relationships with researchers and managers, threatened fisher identities, and poor enforcement capacity. Together, dispute and deeper conflicts limit the effectiveness of singular approaches (e.g., regulation) to mitigate fisher-shark interactions and necessitate multi-pronged solutions with substance-, process-, and relationships-based components. This case study documented one such multi-pronged strategy employing fisher-researcher knowledge exchange, collaborative research, and means of more transparent communication. This strategy has the potential to affect both dispute- and deeper-level outcomes by advancing collective understanding of sharks and shark-handling tools, fisher behavior, and reducing shark mortality. Thus, a sociopolitical approach to problem-solving may yield greater collective benefits to fisheries stakeholders and sharks, with broader implications for the systemic management of complex human and biophysical ecosystem components.

Keywords: human-wildlife conflict, problem definition, sharks, stakeholder engagement, Hawai'i

INTRODUCTION

Natural resource management often employs partial problem framings that favor biophysical, "scientifically objective" information (Young N. et al., 2016; Stephenson et al., 2017). Given stakeholders' diverse perceptions of the problem (Bardwell, 1991; Adams et al., 2003; Ebbin, 2011), deciding whose narratives to include makes the problem definition process inherently political (Weiss, 1989). Moreover, problem framings are often inattentive to the sociopolitical, conflict-laden landscapes within which natural resource management problems manifest and evolve (Nie, 2001). Failed problem-solving endeavors ignoring social, cultural, and political contexts have been documented in conflict resolution (Ginges et al., 2007; May, 2013) and conservation management (Dickman, 2010; Clark and Slocombe, 2011). Even as socioeconomic and biophysical data are integrated to improve management outcomes (Stephenson et al., 2017), using simplified models for decision-making can lead to overly simple solutions (Scott, 1998). It is essential in problem-solving endeavors to embrace the full range of systems-level complexity of natural resource management problems (Ostrom, 2007; Palsson et al., 2013; Aswani et al., 2018).

Sociopolitical analyses have begun to shed light on the systemic complexity of human-wildlife conflict and management. For example, an emerging body of literature describes terrestrial management efforts aimed at biological and tangible problems of human-wildlife interaction that instead exacerbated conflicts. In some cases, management regimes generated symbolic meanings for large terrestrial wildlife among stakeholders who then viewed wildlife as negative representations of state governance (Nie, 2001; Naughton-Treves and Treves, 2005) or the interests of distant, privileged environmental groups (Skogen et al., 2008). Where these symbolic meanings around human-wildlife conflict are ignored, animosity toward wildlife may persist even after negative human-wildlife interactions have been resolved (Dickman, 2010). Other cases point to implementation breakdowns due to failure to account for power dynamics between managers and stakeholders (Webber et al., 2007; Clark and Slocombe, 2011). Strategies that instead engage stakeholders to take stock of and improve relationships among stakeholders and between humans and wildlife may be better positioned to resolve human-wildlife conflict (Marchini et al., 2019). Madden and McQuinn (2014) present two success stories in human-wildlife conflict management where problem-solving processes account for politics and stakeholder relationships. Together, these studies caution against narrow problem framings of human-wildlife conflict and highlight the benefit of considering sociopolitical contexts and engaging stakeholders as we strive toward human-wildlife coexistence.

Sharks (subclass: Elasmobranchii; superorder Selachii) present an opportunity to diversify the human-wildlife conflict literature. Research around human-shark interaction has focused primarily on public (Friedrich et al., 2014; Garla et al., 2015; O'Bryhim and Parsons, 2015; Acuña-Marrero et al., 2018) and fisher (McClellan Press et al., 2016; Drymon and Scyphers, 2017; Shiffman et al., 2017; French et al., 2019) attitudes toward shark conservation, fisheries interaction patterns and their economic and ecological implications (Stevens et al., 2000; Glaus et al., 2019; Mason et al., 2019), and efforts to mitigate shark depredation and bycatch (Carruthers and Neis, 2011; Gilman et al., 2015; Oliver et al., 2015; Kumar et al., 2016; Mitchell et al., 2018). Researchers have also focused on characterizing the global shark seafood trade (Clarke et al., 2006; Shea and To, 2017), shifting livelihoods of shark fishers (Jaiteh et al., 2017), and emerging opportunities and challenges in shark tourism (Techera, 2012; Vianna et al., 2012). This body of work exemplifies the diverse positions from which researchers and managers understand sharks and people (Molony and Thomson, 2020) including public safety, shark conservation, and fisheries impacts. Collins et al. (2020) explore the more intricate relationships between shark management measures' efficacy and diverse fisher economies, adaptive capacities, social norms, and perceptions of management. However, a need for research and policy mechanisms to account for behavioral (Fulton et al., 2011) and socioeconomic factors affecting the feasibility of shark management and bycatch mitigation measures persists (Campbell and Cornwell, 2008; Booth et al., 2019). The sociopolitical lens, with its attention to stakeholder relationships, diverse problem definitions, histories, and power, has received limited attention in human-shark conflict and shark management to date.

This research contributes to the existing body of literature by delivering a localized sociopolitical analysis of fisher-shark interactions. In this West Hawai'i case study we engaged small boat fishers to expand upon biophysical, conservation-driven problem framings around fisher-shark interactions and account for stakeholder conflicts and power dynamics. An adapted theoretical framework connects concepts of problem definition and layered conflict, providing structure to examine complex problems and comment on the equity and effectiveness of potential solutions. Coupled with its stakeholder interview and observation methods, this research examines the relevance of stakeholder conflict, power, history, and identity to fisher-shark interactions and the importance of process and relationships in reconciling them.

We explore problem-solving in fisheries through two overarching research questions: (1) What layered conflicts are embedded in fishers' definitions of the fisher-shark interaction "problem"? and (2) How might solutions address both dispute and deeper conflicts for the benefit of stakeholders and sharks?

METHODS

Theoretical Framework

This study adapts Madden and McQuinn (2014) conflict framework to examine problem-solving in fisheries. Madden and McQuinn draw from two conflict models. The first identifies three Levels of Conflict (Canadian Institute for Conflict Resolution, 2000), in descending order: dispute, underlying conflict, and identity-based conflict. Dispute encompasses the presenting, often tangible conflict, such as the contested boundaries of a marine protected area. Underlying conflict provides relational and historical context for the dispute and

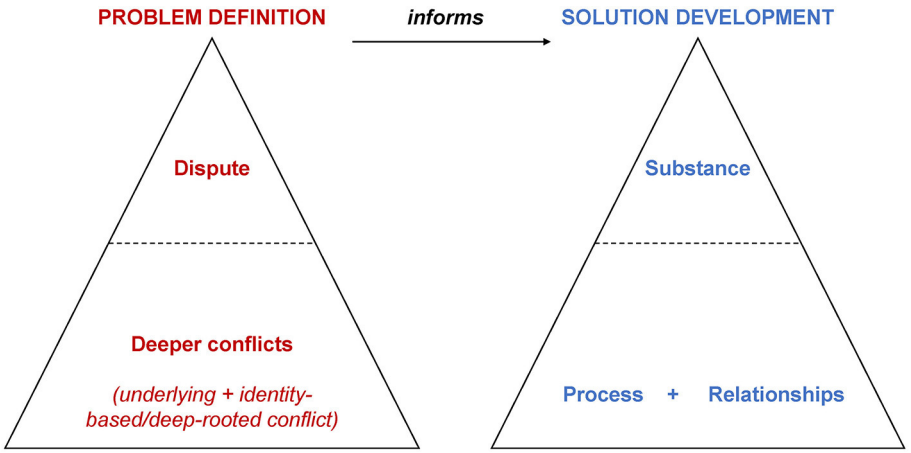


FIGURE 1 | Our theoretical framework examines the process of problem definition, which in turn informs developing solutions in resource management. The framework is adapted from Madden and McQuinn (2014), who mobilized the Levels of Conflict (Canadian Institute for Conflict Resolution, 2000) and Conflict Intervention Triangle models (Moore, 1986; Walker and Daniels, 1997, p. 22).

is often reflective of actors' past interactions and unresolved conflicts that may or may not be related to the dispute at-hand. How actors navigate the dispute at-hand can be deeply affected by underlying conflict. An example of underlying conflict is persisting mistrust between actors based on past management decisions. Identity-based, or deep-rooted conflicts, derive from perceived threats to actor values, culture, and identity. Identity-based conflict might arise, for example, as resource users perceive management as a threat to their autonomy.

In Madden and McQuinn's (2014) Conflict Intervention Triangle—adapted from Moore (1986) and Walker and Daniels (1997)—substance comprises the triangle's apex, and process and relationships sit at its basal corners. Process is defined as, "decision-making design, equity and authority, and how (and by whom) these are exercised" (Madden and McQuinn, 2014, p. 102). Relationships refer to those between individual actors or stakeholder groups and the levels of trust and respect entwined in them. While substance, process, and relationships are all connected and integral to conflict resolution, substantive interventions are considered better suited for dispute, and relationship- and process-based interventions better suited for underlying and deep-rooted conflicts.

Researchers have applied Madden and McQuinn's framework to examine the social conflicts that underlie human-wildlife conflicts, highlighting their relevance to management (Dorresteijn et al., 2016; Hill, 2017; Crespin and Simonetti, 2019). Our study adapts the Madden and McQuinn framework to investigate stakeholder perceptions of how problems are defined and solutions are developed. We adapt their conflict models to interpret these two critical processes in fisheries management (**Figure 1**). Madden and McQuinn (2014) framework depicts three levels of conflict (dispute, underlying, and identity-based) and corresponding conflict resolution approaches (settlement, resolution, and reconciliation). Zimmermann et al. (2020) build on this framework and provide additional guidance, including specific approaches to identify and address each level of conflict. However, underlying and identity-based conflicts are often intertwined, and both derive potential benefits from reconciliatory solutions (Lederach, 1997; Lundy and McGovern, 2008). Within our study context, separating underlying and identity-based conflicts provides little benefit to our problem-solution analysis. We therefore interpret underlying and deep-rooted conflict together as facets of "deeper conflict" rather than as distinct tiers of a conflict hierarchy that benefit from distinct solutions. Further, we demonstrate interactions between dispute and deeper conflicts, which necessitate solutions incorporating dimensions of process, relationships, and power.

Case Study Site and Problem

In December 2016, the National Marine Fisheries Service (NMFS) proposed a rule to list the oceanic whitetip shark (*Carcharhinus longimanus*) under the Endangered Species Act (ESA), citing significant declines in abundance throughout its habitat range due to overexploitation (Young C. N. et al., 2016). Subsequent management measures would likely focus on pelagic high-seas fisheries that inflict high shark bycatch and mortality rates (Bonfil, 1994; Gilman et al., 2008). However, the ESA listing's undetermined status and managers' and researchers' growing interest in oceanic whitetip sharks provided an opportunity to explore fisher-shark interactions in Hawai'i more broadly. We took this opportunity to conduct a sociopolitical examination of the West Hawai'i small boat fleet and its interactions with pelagic sharks, including, but not limited to the oceanic whitetip shark.

Information around pelagic shark interactions within the West Hawai'i small boat fleet is largely undocumented. Shark species, interaction frequencies, outcomes, and determining factors for fisher and shark behavior were among the unknowns. Prior to this study, anecdotal evidence indicated that fisher-shark interactions within this fishery could produce negative outcomes for both fishers and sharks; namely, loss of fisher catch and gear, and shark injury. Fishers also expressed a desire to reduce

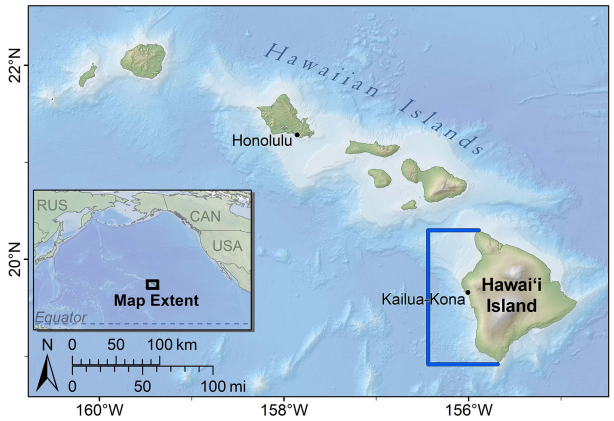


FIGURE 2 | Map of Hawai'i Island within the main Hawaiian Islands. The West Hawai'i region (including the location of Kailua-Kona) is indicated in blue.

these interactions, which they described as largely incidental and undesirable. Thus, shark conservation, research, and fishery impacts perspectives provided rich context to examine the implications of problem definition for developing solutions.

West Hawai'i refers to the leeward, western coast of Hawai'i Island (**Figure 2**). Its calm waters and proximity to pelagic species allow fishers to accumulate a relatively large number of fishing days per year with good visibility for pelagic shark observations. The West Hawai'i small boat fishing community represents diverse fishing methods, experience levels, and ethnicities. It includes perspectives from local, outer island, and continental U.S. fishing cultures, and membership from the recreational, part- and full-time commercial, and charter fisheries.

West Hawai'i fishers' exposure to fisheries management and research also lends itself well to a sociopolitical analysis. The West Hawai'i Regional Fishery Management Area (WHRFMA) encompasses four Marine Life Conservation Districts and seven Fisheries Management Areas, with bans on SCUBA spearfishing and the take of reef sharks and rays (State of Hawai'i Division of Aquatic Resources, 2019). Aquarium fishing has been suspended since 2017, with a recent environmental impact statement requesting limited permits rejected by the Board of Land and Natural Resources (BLNR votes 7-0 against environmental impact statement for aquarium fishing permits in West Hawaii, 2020). Several local and international non-governmental organizations (NGOs) operate there in lobbying and research support capacities (Tissot et al., 2009). West Hawai'i is also a focus area for the National Oceanic and Atmospheric Administration's (NOAA) Habitat Blueprint and Pacific Islands Ocean Observing System (PacIOOS) Hawaiian Islands Sentinel Site Cooperative. All of these inform the West Hawai'i fishing community's perspectives on local science and management.

In months following the oceanic whitetip shark's proposed ESA listing, a team of University of Hawai'i and NMFS-affiliated shark researchers expanded its pelagic shark-tagging efforts to include West Hawai'i small boat fishers. The team hoped that fisher participation would enhance understanding of sharks' movements and alternatives to shark-handling practices that could result in mortality. The team trained fishers in shark-tagging protocols and shared information about shark life history, vulnerability to fishing activity, and management measures. Financial incentives were awarded to fishers who deployed electronic and identification tags on oceanic whitetip, silky (*Carcharhinus falciformis*), and bigeye thresher (*Alopias superciliosus*) sharks. The tagging program prioritized these species given their listing in the Convention on International Trade in Endagered Species' (CITES) Appendix II, and the International Union for Conservation of Nature's (IUCN) Red List of Threatened Species as "Critically Endangered" and "Vulnerable." The early growth of this collaborative program, now termed the Hawai'i Community Tagging Program (HCTP), provided a valuable opportunity to observe evolving fisher-researcher relationships with a substantive focus on fisher-shark interactions.

Data Collection

Across disciplines, stakeholder perspectives have revealed new technical problem framings (Leong et al., 2007) and clarified the relevance of politics, stakeholder values, and culture to solution design (Watkin et al., 2012; Madden and McQuinn, 2014). Participatory processes have also been enlisted to challenge the biophysical constraints of fisheries problem-solving (Mikalsen and Jentoft, 2001; Beierle, 2002; Ebbin, 2011; Sayce et al., 2013). We combined semi-structured interviews with participant observation to explore problem definition and potential solutions through fisher perspective and fisher-researcher relationships. We sought to generate new understanding from the data rather than interpreting data through pre-established hypotheses using a qualitative, inductive approach. IRB clearance was obtained through NOAA's Pacific Islands Fisheries Science Center (PIFSC) under Joint Institute of Marine and Atmospheric Research (JIMAR) exempt project 19449, Socioeconomics of Western Pacific Fisheries.

Data were collected primarily through semi-structured interviews. Preliminary meetings with members of local fisher-oriented NGOs, HCTP, fishers, and fisheries social scientists informed development of an interview guide, solicited guidance around best practices for fisher engagement, and identified contacts within the West Hawai'i fishing community. Public workshops were co-hosted by the lead author and HCTP given their shared target audience. Participation was also advertised through flyers distributed in tackle shops and harbors, and announcements in the local Hawai'i Fishing News magazine. These printed materials were unsuccessful in connecting the lead author to new research participants. Additional interviewees were identified through the snowball sampling method, which relies on established participants' referrals to identify new contacts and user groups in their community (Atkinson and Flint, 2001).

In addition to serving as a subject of this case study's observations, the HCTP and its public workshops provided a venue for the lead author to promote participation in and share results from this case study. These shared venues for the lead author and HCTP to recruit participants and communicate

results may have produced some bias among interviewees who also participated in the HCTP. However, data collected for this project are distinct from that of the HCTP and fewer than half of this study's interviewees were HCTP participants.

The fisher engagement strategies used in this case study and the HCTP had some similarities. Both initiatives sought to accommodate fishers while operating within the study scope and budget. Outreach efforts and public workshops were centered around Kailua-Kona, where many participants either lived or worked. Workshops were scheduled after consultation with research participants in an effort to make them more accessible to the local fishing community. For individual and small group meetings, HCTP PI and the lead author for this case study met with fishers in settings of their choice and made travel arrangements to accommodate fishers' schedules as much as possible. Case study interviews were conducted in family homes, at the Honokōhau Harbor, and at local restaurants. Informal, fisher-led HCTP gatherings also promoted participation in the tagging program. For example, one fisher hosted a gathering in his home. Previously the HCTP had difficulty connecting with commercial fishers given their unpredictable and demanding fishing schedules.

Case study interviews were conducted in-person from September 2017 to June 2018, and typically lasted 1–3 h. We asked participants about their relationship to fishing and fishing history; information sharing in Hawai'i Island fisheries; shark interactions and handling practices; and local fisheries management and science (see Appendix in **Supplementary Material** for detailed interview guide). Interviews continued until data became saturated–meaning that new interviews repeated themes and ideas captured in former interviews (Saunders et al., 2018)–resulting in a total of 29 interviewees. Interviews were audio recorded and detailed notes were written up as soon after the interview as possible. Interviews were transcribed and interviewees were given a copy of their interview transcripts for voluntary review and revision.

Observations supplemented interview data on fishing practices and fisher-researcher interactions. The lead author conducted participant observation on three occasions with the HCTP research team. These included a chartered shark-tagging trip on a commercial fishing vessel out of Kailua-Kona and two public shark-tagging workshops held in October 2017 and 2018. At both workshops, the HCTP team trained fishers in tagging protocol, distributed shark-tagging gear, and reported on the progress of their research.

Data Analysis

Data analysis followed an inductive process typical of grounded theory (Glaser and Strauss, 1967). Data were collected and primary analyses completed before selecting a theoretical framework. This approach enabled pursuit of themes important to interviewees and elicited connections between relevant biological, social, economic, and political components. The lead author coded all interview transcripts using NVivo software (NVivo, RRID:SCR_014802, version 11 Plus). Preliminary codes followed the interview guide's broad themes: fisher identity, sharks, information-sharing, and management. Additional codes were created liberally to capture all relevant information, including descriptions of West Hawai'i fishing practices and cultures through time, economic context, fishing motivations, human well-being, power, and knowledge. A transcript excerpt could be coded for multiple themes, overlapping or separate across the text. Two rounds of coding ensured that all relevant data were represented in the codebook and that codes were organized to reflect thematic relationships and consolidate coding redundancies.

In February 2019, the lead author returned to the study site to present preliminary results to research participants at a public HCTP meeting and solicit feedback. Twelve of ∼30 attendees were interviewees. Although interviewees did not volunteer feedback in front of the group, the few who later corresponded with the lead investigator shared positive comments about the way their input was represented and volunteered their support for future collaborative research endeavors.

TABLE 1 | Interviewee demographics.

Characteristic		*n*
Age, years	18–35	4
	36–50	7
	51–65	12
	66–80	6
Ethnicity	Caucasian	11
	Asian[b]	11
	Hawaiian	3
	Hawaiian-mixed	4
Origin	Hawai'i Island	14
	Neighbor islands	7
	Continental US	8
Type of fishing	Commercial only[a]	12
	Charter and commercial	5
	Charter only	4
	Recreational/retired	8
Years fished in Hawai'i	5–15	3
	16–30	10
	31–45	10
	46–60	6

[a] *3 of 12 commercial only participants identified as full-time commercial fishers.*
[b] *8 of 11 Asian participants were Japanese.*

Research Participants

The 29 male interviewees represented diverse demographic and fishing identities (**Table 1**). Interviewees averaged 30 years of fishing experience in West Hawai'i per fisher; a conservative estimate, excluding shoreline fishing that predates boat fishing ventures or formal fishing careers, and rich fishing experiences inherited from generations past. Independent of their charter, commercial, and recreational identities, a total of 16 interviewees described non-fishing occupations that either supplement their fishing income or serve as their full-time position. On average, interviewees described fishing for over 160 days per year in the peak of their careers. Interviewees' participation in collaborative

research and management-related fisher engagement also varied, with most having limited experiences in either.

Although participants' repertoire of fishing methods was extensive, those described most frequently by interviewees were handlining ($n = 23$), trolling ($n = 21$), and fishing with live bait (n = 18). Interviews covered a diverse range of target species, the most popular of which were deep bottomfish (primarily snappers and a grouper), ʻahi (either bigeye or yellowfin, *Thunnus obesus* and *T. albacares*, respectively), marlin, and ʻōpelu (mackerel scad, *Decapterus macarellus*). Other target species cited in interviewees' primary fisheries, past and present, included pelagics like mahimahi (*Coryphaena hippurus*), aku (skipjack tuna, *Katsuwonus pelamis*), and ono (wahoo, *Acanthocybium solandri*), reef fish both for consumption and sale in the tropical fish trade, Kona crab (*Ranina ranina*), and black coral.

RESULTS

Interview and observational data revealed dispute and deeper conflicts framing the fisher-shark interaction problem, along with several substantive, process-, and relationship-based solutions.

Problem Definition

We found that fisher-shark interaction problem framings are both disputative and steeped in deeper levels of conflict related to fisheries science and management. We expand on both of these below.

Dispute

Shark attributes described by interviewees were coded as negative, positive, or neutral depending on their overall cost or benefit to fishing activity. Negative descriptors were the most prominent (49% of all references). Most commonly, interviewees described sharks as competitors (26% of all references), generally and through two specific mechanisms. They described sharks as either depredating–removing hooked fish or bait from fishing gear (Gilman et al., 2008)–or deterring target fish from interacting with their gear. Competition for fish, in turn, translated to competition for fisher income and a threat to fisher livelihoods.

Sharks' competitive impact on fishing was greater for those who rely more on landing fish for income—full-time commercial fishers, for example, as opposed to part-time commercial, charter, or recreational fishers. Despite occupational differences, interviewees described these consistencies across the West Hawaiʻi small boat fleet: increasing costs of fishing material, uncertain landings, and a fishing community that has grown rapidly in recent years. These conditions increase competition for fishing spots and a decent price at which to sell catch. More than two-thirds of interviewees described fishing as a "lifestyle" associated with financial insecurity.

Interviewees described sharks differently according to species. We selected three species that illustrate this diversity: tiger (*Galeocerdo cuvier*), oceanic whitetip, and thresher sharks (*Alopias* spp.) (**Figure 3**). Some descriptors were linked to interviewees' shark interaction frequencies and handling practices. Shark aggression and depredation (**Figures 3A,B**) were often talked about together and associated in some cases with shark mortality for its burden to fishing activity. "An aggressive, hungry shark is probably gonna die," said one fisher. Sharks' role as fish indicators (**Figures 3B,C**) may increase interaction frequency because, "If there's sharks around, then you know there's fish around, why leave?" However, sharks as fish indicators were categorized positively because, as one fisher noted, "It's a good sign too. When you're getting the interactions with the oceanic whitetips there's more fish around normally." In contrast, the danger (**Figure 3A**) or hassle (**Figure 3C**) that a shark imposes on a fisher might diminish fisher willingness to interact with certain species. Importantly, the relationship between shark descriptor and fisher behavior is not always the same. For example, tiger and thresher sharks were both described as economically valuable. However, tiger sharks' economic value was attributed to the benefit of their sensationalized image to the tourism industry, while thresher sharks' derived from their market value. Landed thresher sharks thus provide fishers opportunity for direct financial compensation, while tiger sharks' economic value is relatively inconsequential for fishers.

Interviewees noted a number of factors related to shark attributes, landing opportunity, social pressure, physical capacity, and investments in time and finances that affect their decisions during a shark interaction (**Table 2**). Any number of these can play a role in fisher behavior during a shark encounter, but the cost-benefit calculus varied by individual. For example, considering how many sharks vs. target species are in the area, one fisher commented:

> [If] it's just nothing but sharks… [that's] time to quit, because not only are we going in the hole with our gas and our ice and our bait, they're taking our tackle, destroying our stuff.… We got a thousand dollars in the hole, we just have to let the conditions change out there until those damn sharks move out of here.

Another fisher said, "If there's a lot of tuna and a lot of sharks, you find different ways to kind of get around the sharks." At the intersection of shark species and financial considerations lies the market value of mako (*Isurus* spp.) and thresher sharks. One fisher commented: "It's really bycatch. You're going for ʻahi and all of a sudden a thresher bites, and then you look at this thing, you don't have anything in your box, you go, 'Oh I can make money killing this shark.'" Some fishers noted social factors affecting their behavior: "You don't know who's in the other boat too, so you [don't want to] just shoot [the sharks]."

Despite sharks' competitive impacts and the physical, financial, and time investment risks they pose to fishers, many interviewees described not devoting much thought to sharks prior to engaging in this case study or the HCTP. Many interviewees ($n = 17$) described sharks as incidental, non-target species. Fishers thus dedicated relatively little observational attention to sharks and discussed them with others in the fishing community only peripherally to their main fishing activity.

Deeper Conflicts

Deeper conflicts derive from threats to identity, culture, and values, or unresolved conflicts between actors. The

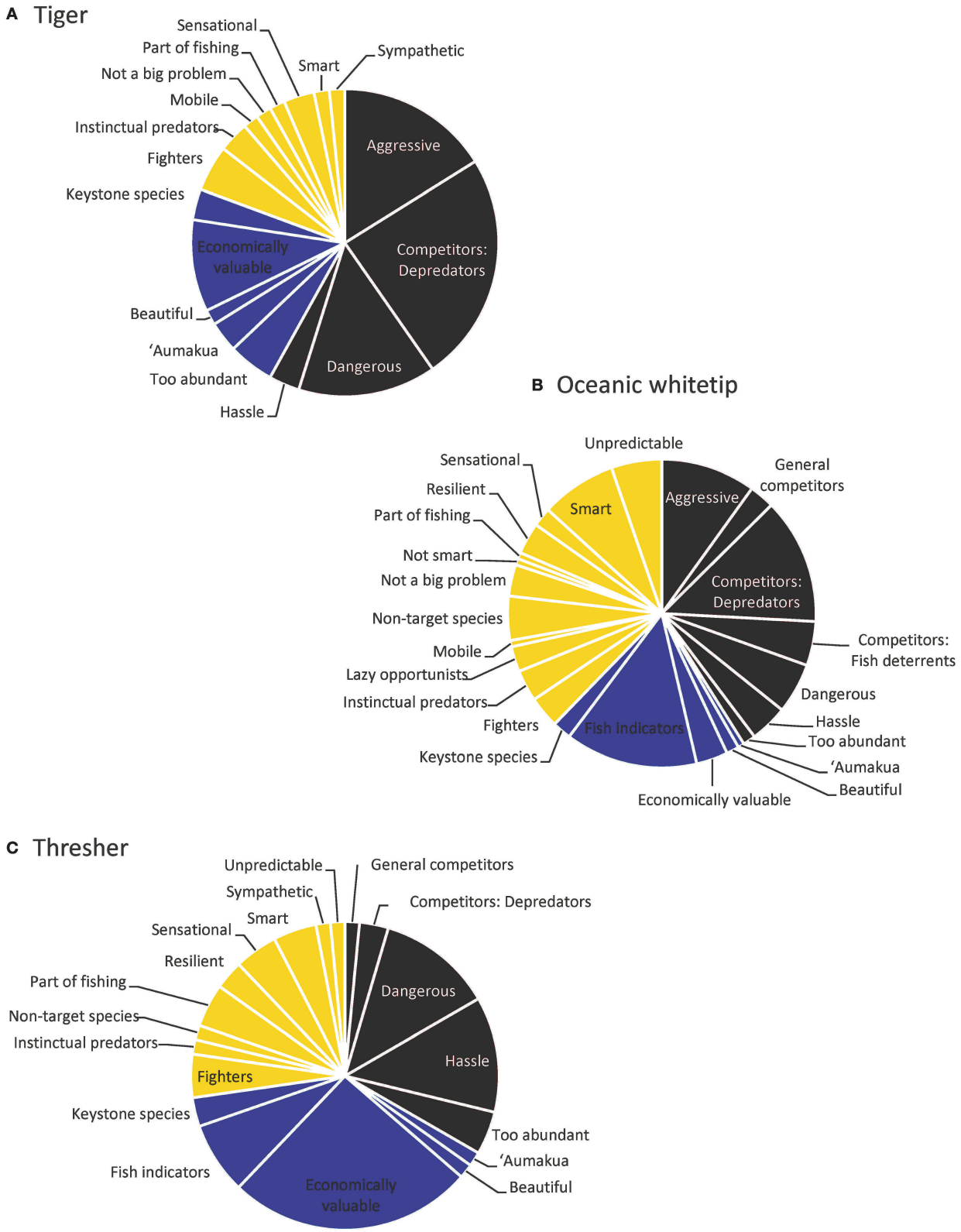


FIGURE 3 | Interviewees' negative (dark red), positive (blue), and neutral (beige) descriptors of **(A)** tiger, **(B)** oceanic whitetip, and **(C)** thresher sharks, by number of references across all interviews. See **Supplementary Table 1** for definitions and examples of all shark descriptors.

TABLE 2 | Decision-making factors during a shark interaction.

Factor category	Factor (# of interviewees citing each)	Effect on fisher behavior; Illustrative quote(s)
Shark attributes	Shark accessibility (20+)	The degree of physical access a fisher has to a shark determines his/her behavioral options. Namely, whether or not the shark is hooked or at the surface.
	Shark persistence (19)	Shark persistence despite fisher handling increases the readiness of fishers to apply more severe handling practices. Tiger, blue (*Prionace glauca*), and oceanic whitetip sharks were among those described as persistent.
	Number of sharks (16)	Coupled with shark persistence and aggression, a high number of sharks may result in fishers leaving an area.
	Survivorship (12)	Some fishers described their shark-handling preferences based on the perception that they do not result in shark mortality or significantly impact shark populations.
Landing opportunity	Shark market value (15)	The market value of mako and thresher sharks offers fishers the added opportunity to land them for sale.
	Target species' presence (15)	If target species are present, a fisher is less likely to leave and more likely to attempt to fish around or handle a shark.
	Fish on the line (6)	If a fish is on the line, fishers may be more receptive to short-term strategies that are otherwise unattractive (e.g., shark feeding, jugging).
	Time of day (5)	Small windows of opportunity for fish bites make fisher decisions more critical and reduce behavioral options.
Social pressure		The increased likelihood of being observed in daylight may also restrict fishers' handling options if they may be perceived as socially undesirable.
		If [the sharks] come and get you at prime time, you're done…. Dusk or dawn, yeah. You see the first crack of gray…. Our movements, the way we chum, the way we check our baits, becomes ten times as critical.
	Other boats (10)	Presence of other boats in an area may discourage fishers from using certain shark-handling practices, redistribute shark impacts among fishers, or inhibit a fisher's ability to move given already occupied fishing spots.
		When there's a lot of charter boats out there live baiting… Then there's less shark predation on my side.
Physical capacity	Safety (14)	Shark-handling is a physically demanding activity. Tools can reduce its physical stresses, but also pose additional bodily risks. Safety considerations' impact to each fisher's behavior varies according to personal preference, physical ability, age, and gear/vessel configuration.
	Gear (10)	Fishers' typical gear configurations are limited in the access to sharks and handling practices they enable.
		When we go out for fishing, we're just rigged for fishing…. So you kind of use what you got, and what you got to work with.
	Crew (6)	More hands on deck make physically challenging handling practices more accessible to some fishers. It may also discourage the use of certain tools (e.g., guns) for safety reasons.
	Vessel size (4)	Fishers with larger vessels have access to more behavioral options and are able to handle larger sharks.
Time/financial investment	Distance traveled (2)	Fishers may be disinclined to travel to distant fishing grounds if they know there are sharks in the area. Fishers may also consider a wider range of behavioral options if they are already fishing a distant area.

Direct quotes in italics.

deeper conflicts that emerged from interviews drew from broader discussions of local fisheries management and fishers' experiences with researchers and managers. Interviewees noted several problems in local fisheries management and science (**Table 3**). The most commonly cited were disconnect in fisher-manager or -researcher logic, experience, and power; misplaced management focus on small boat fishers; lack of enforcement; politicized decision-making; questionable data validity; and a lack of transparency around science and management. Each of these issues degraded fishers' perceptions of the legitimacy of fisheries management and science (noted with asterisks in **Table 3**). Interviewees referenced, for example, the scapegoating of fishers for less visible or manageable issues: "Most of these laws are people bored and they wanna blame fisheries for the depletion of fish, or hunters for depletion of animals in the forest, even though they don't see the real issue."

Often, deeper conflicts came to light as fishers described engaging with researchers and managers. Three problem themes emerged in direct connection with these experiences: fishers' voice, apprehension around engagement, and resignation from engagement processes (**Table 4**). Although interviewees

TABLE 3 | Problems in fisheries management and science described by interviewees.

Problem theme (# of interviewees citing each)	Included perceptions of…	Illustrative quote(s)
Stakeholder disconnect* (17)	Disconnect between fishers and managers/scientists, often between fishers' on-the-water experiences and manager/scientist logic. Also, a lack of consideration or empathy for how management and research decisions affect fishers.	*Go to the fishermen that are in the water and actually interact with the animals every day. Ask them, first. Before you go to Land Board, all those other people that think they know what they're doing. If people that generally made laws could do that, I think it would open their eyes a lot more as to what actually goes on. Instead of just reading what is on the piece of paper… and signing it off.*
Relative impact* (14)	Misplaced focus of research or management. Often related to perceptions of other fishing groups having greater resource impacts and/or lesser regulatory oversight.	*Sportfishing and the local commercial fishermen are minute compared to the big corporation or big fishing companies in the state of Hawai'i. The longline fleet takes top priority… And they get away with a lot more than anybody else could, and if the smaller fishermen came in there to voice what they thought about the tuna… they just went in one ear out the other ear.*
Lack of enforcement* (14)	A lack of enforcement in terms of capacity as well as practice.	*They have enough [state enforcement] people on this island to do one 8-h shift 5 days a week. For the whole island, from the top of the mountain to 3 miles offshore.*
Politicized decision-making* (13)	Management and regulations based on public sentiment and special interests rather than science and rationality. Also, science used to support political decisions.	*Unfortunately a lot of regulations are made not by science but by emotion.* *Just like every scientist I know [does], they only take the information that proves [their] fact…. Every single scientist.*
Data validity* (11)	Questionable validity of data collected for fisheries management, its use in decision-making, and the ways that might be improved.	*I'm all for proper management if I can see the results. Show us where those numbers came from.* *The fishermen are out there all the time. They're out there in fact more than the scientists I think, in numbers. So they can be an asset.*
Transparency* (8)	A lack of transparency or clarity with regard to managers' or scientists' motives and goals.	*All we know is that you guys just want us to try and tag [sharks]. And that they may be on the endangered species list…. What more are you looking for? What's your objective? What's your goal?* *I stood up, I said, "How did you get that blue line since it's not reported?" And [scientist] says, "We have our ways."*
Lack of compromise (8)	Management discourse and processes being biased and unwilling to compromise or consider other perspectives.	*You've got the total left that just want regulation… and then you've got the other side that is just all or nothing. There's nothing in the middle…. There's no management.* *A lotta time the decision is already made and they just have these public hearings and all these things… It's so one-sided that it just [always goes] one way already.*
Permanence (7)	Management measures as permanent and non-adaptive.	*They had that 10-year ban in Ka'ūpūlehu, that thing is never gonna open…. it's never gonna have a review after 5 years. It's because the state [doesn't have] any money.* *If you make it a law that you cannot kill this… now you're going to get a million sharks around you, you can't even fish. There's gotta be a balance…. Because in the future you might not be able to retract that law.*
Inequity* (5)	Inconsistencies in management decision-making, which affords benefits to certain groups and targets others disproportionately for regulation.	*It really seems like they pick and choose… What rules they want to push, what rules they want to enforce, to kind of pick on a specific group of people.*

*Direct quotes in italics. *Issues that degrade fishers' perceptions of the legitimacy of fisheries management and science.*

expressed interest in sharing their voice and perspectives with researchers and managers, opportunities to do so were often described as limited or superficial. Several interviewees described past experiences that resulted in mistrust of managers and researchers, fear of losing fishing access, or a withdrawal from fisher engagement processes altogether. In some cases fishers expressed these concerns, facilitating transparent fisher-researcher discussions that helped to overcome them:

TABLE 4 | Interviewee perceptions around fisher engagement.

Engagement theme (# of interviewees citing each)	Included perceptions of…	Illustrative quote(s)
Fishers' voice (19)	A need for fishers' voices to be heard by fisheries management actors and to affect meaningful change. Often this was a fisher goal that went unmet during engagement.	*We have no voice. The fishermen have no voice.* *You guys are probably gonna be fisheries managers or advising fisheries managers and stuff, and at least you listen.*
Apprehension (6)	Fishers' concerns for or fears related to fisher engagement motives and outcomes.	*That's where everybody shuts up… 'cause [things] end up out of our control. And then next thing you know it's a law, and we can't go near them, or we can't fish these areas.*
Resignation (5)	Fishers giving up on engagement opportunities given their past experiences.	*I told him, "No dude, I'm done with that kinda deal."… At the end, I felt like it was so much effort coming from our side, with no end result. Or meaningless time that we spent there… No matter what we say or do, there's gonna be no results.* *I go off and on, but not taking interest like I used to because it doesn't matter. That's the sad part.… Why have people go over there and have issues where somebody really cares about something, voices their opinion, and [it doesn't] matter?*

Direct quotes in italics.

> I had to ask about your goals and intent because… I go to meetings now, I know what they're trying to do to Kona. They're trying to make this an aquarium.… They have to also think about the culture. And the local people here.

In discussions of fisheries management and engagement, interviewees perceived certain actors and knowledge types to wield greater power than others. Often, interviewees described themselves or their communities as adversely affected by these power dynamics: "They make these decisions for this stuff without really knowing the impact… it has on our life." Interviewees described financial capital and formal, academic, and scientific knowledge to facilitate access to management decision-making and leverage over its outcomes. One fisher advised, "To fix all the problems, you have to get your degree… and find [a] route to the money." Fishers, who offered experiential knowledge through communication styles often considered informal, felt disadvantaged by this dynamic. Interviewees perceived managers to assert their power through uninformed decision-making, evading answers when fishers' knowledge challenged their own, and hosting public meetings with limited outreach or accessibility. Interviewees perceived researchers' power through their representation of fisheries and delivery of fisher data to managers, and a lack of transparency around their science or goals. Interviewees felt both researchers and managers influence decisions with relatively little impact to themselves compared to fishers who experience direct impact. Some research participants expressed concerns that this case study and the HCTP intended to create new fishing regulations and inquired about funding sources. These examples of power embedded in deeper conflict cut across various problem themes from **Tables 3**, **4**. Financial capital and influence on public perception also emerged from interviews as forms of power wielded by the tourism industry, high seas fisheries, environmentalists, the wealthy, and NGOs.

Many of these deeper conflicts denied or threatened key elements of fisher identity. Interviewees most commonly discussed their identities as self-reliant, stewards, local, multi-generational fishers, and scientifically curious. These emphasized fisher agency, practices of self-management to sustain fisheries, and commitment to place and local fishing culture. Interviewees also took pride in their acuity for on-the-water observation and experimentation and expressed interest in novel scientific data collection methods and projects, including shark-tagging data.

Solution Development

This case study documented several solutions rooted in substance, relationships, and process. These were either discussed by interviewees or exemplified in the engagement strategies employed by the lead author or HCTP.

Substance

The substantive solutions that emerged from interviews and observations included shark-handling alternatives, regulation, information provision, and financial incentives. **Table 5** provides fisher perspectives and exemplary quotes for each of these solution types. Fishers were amenable to shark- handling alternatives that would preserve factors like their landing opportunity and safety (**Table 2**). For example, "jugging" consists of rigging a jug or floated object to a baited hook. This contraption is often deployed unattached to the vessel. Once taken by a shark, it maintains the shark's surface position and deters it from the fishing area or target species. Research participants raised such ideas as biodegradable jugs or more

TABLE 5 | Substantive solutions.

Solution element (# of interviewees citing each)	This study documented…	Illustrative quote(s)
Shark-handling alternatives (28)	Firearms, releasing hooked animals, jugging, and shark avoidance strategies were among the most commonly discussed shark-handling alternatives. Preferred strategies varied by individual, but were often viewed as the most efficient, safest, or only available options to protect fishing opportunity. Fishers were receptive to the idea of novel shark-handling alternatives.	*A lot of times as a fisherman it was hard because if you could wave the magic wand, please go away, you would.* *I don't know what else could be done… That's the only thing I would listen to, if you had a deterrent. Other than that I wouldn't go to listen to anything else.*
Regulation (26)	Discussion of various regulatory tools, including area closures, shark finning bans, fishing licenses, and catch limits. Interviewees linked poor enforcement with perceptions of management as ineffective and illegitimate.	*A law with no enforcement is merely a suggestion… Over here, there's zero enforcement.* *In Hawai'i, it's all about who you know, not what you know. In Alaska, it doesn't matter. The rules [are] the rules*.
Information provision (24)	New information about sharks' cultural significance or biology having limited potential to change fisher behavior and reduce shark mortality.	*The Hawaiians said release all the sharks that you catch because it was their cultural practice or something. So, I just… let them go.*
Financial incentives (15)	The opportunity for financial incentives to address the costliness of shark interactions. By offsetting costs and acknowledging fishers' livelihood insecurities, financial compensation may legitimize and increase fisher access to a broader number of behavioral and shark-handling options.	*You get the grant, and there has to be some type of reward… "Cause if not, everything comes down to the end of the month. You pay your mortgage, you pay your college loan….* *You give me x amount of dollars to go tag every single shark that comes by the boat, they'll live. I'll spend all day tagging sharks*.

Direct quotes in italics.

readily eroding jugging rigs, and provided anecdotal evidence of shark-tagging as an effective shark deterrent. Interviewees also discussed various regulatory measures, but often in tandem with a lack of enforcement or legitimacy that cripples their effectiveness. Some fishers requested informational tools, like shark identification guides with species' protected statuses. On rare occasions, fishers described modifying their shark-handling practices as they acquired new information. One fisher said after learning of thresher sharks' vulnerability: "Now I will not shoot a thresher shark that I catch. Because [HCTP PI] informed me.… And we have a deep respect for the ocean." But, as one fisher put it, "Where there's sharks there's fish." Challenges inherent in fishing like sharks and financial cost limit fishers' capacity to modify their behavior. By directly offsetting cost, financial incentives may provide fishers access to alternative shark-handling practices like tagging: "You give me x amount of dollars to go tag every single shark that comes by the boat, they'll live. I'll spend all day tagging sharks."

We found that the impact of any substantive solution on fisher perspective and behavior depends on a number of variables including livelihood dependence on fishing and fishing method. A fisher using live bait, for example, has more behavior change flexibility than a handline fisher. Upon encountering a shark, one live bait fisher described his options to continue moving through the area or reduce the likelihood of shark interactions by switching to an artificial lure. Handline fishers, in contrast, are committed to a fishing spot and their chances of landing a fish depend on consistent chumming. This increases the likelihood of shark interactions and makes avoidance maneuvers challenging. Other variables related to fishers' capacity for behavior change are less tangible. Some interviewees, for example, described their receptivity to modify shark-handling practices as a function of their age or experience level.

Process and Relationships

Drawing upon our adapted theoretical framework, we posit that process- and relationship-based solutions may be better equipped to address deeper conflicts. In their discussions of fisheries management, science, and fisher engagement, interviewees described several solution elements attentive to process and relationships (**Table 6**). Interviewees highlighted the importance and inseparability of trust-building and transparent fisher-researcher and -manager communication, connecting these processes to tangible benefits for data collection and scientific knowledge. Fishers also noted that convenience can help to facilitate fisher participation given fishers' demanding schedules and geographic spread. One participant said, "Fishing advocate[s]… They're kinda retired and they have time to make a difference." Finally, several interviewees highlighted the fishery's diversity and the need to account for it in our engagement processes:

> People have all kinds of different perspectives.… I know people that have a high paying job, they fish on weekends, they only catch for recreation.… You gotta get everybody's opinion.… Not only one side of the story. Please capture everybody.

Cross-Cutting Strategies

This case study documented two strategies that cross-cut multiple substantive and deeper-level solutions: collaborative research and knowledge exchange.

TABLE 6 | Process- and relationships-based solution elements.

Solution element (# of interviewees citing each)	This study documented…	Illustrative quote(s)
Communication (23) + Trust-building (16)	The value of two-way communication in building trust and relationships between fishers and researchers or managers; often described together. Interviewees noted the role of key actors and non-threatening approaches in communication and trust-building, and the potential for improved communication to benefit fisher participation and buy-in (e.g., sharing research results with fishers may benefit collaborative data collection).	*That's the reason why nobody [submits data], you know what I mean? But it's a detriment to the research, because if they had … just a short thing they can send to the fishing public, so that they know that all this reporting was not done in vain. But right now, it's a bottomless pit…. You don't know where that information is going.* *Starting a conversation… [don't] just say, "Hey, I'm here, I'm a scientist… where'd you get that and how was the current?" Like, "Hey brah, how'd you do today?" and just feel them out… Not come from the top and nīele[a] and just sneak up on them.*
Convenience (12)	Issues of convenience and accessibility related to fisher engagement. Fishers noted the investment in time and money required to attend meetings and share their perspectives (e.g., paying their way to outer islands or rearranging fishing/livelihood schedules to meet researchers' and managers' needs). Also, challenges engaging with geographically dispersed communities like that of West Hawai'i.	*It's hard to get those guys…. [They're] so spread out, they don't wanna come to meetings. … You get fishermen from way south, and you're going to hold one meeting up here…. If people [are] right here they're going to go right here. But people far away, they're not going to attend one meeting.*
Inclusion (8)	Diverse fisher perspectives varying according to age, experience level, occupation, fishing method, and geographic origin. These require researcher and manager efforts to account for diversity. Fishers also noted differences across the island chain.	*Get the old-time fishermen, the new fishermans, and they probably all got different opinions.* *People have all kinds of different perspectives too. 'Cause I know people that have a high paying job, they fish on weekends, they only catch for recreation…. You gotta get everybody's opinion, please…. Make sure now! Not only one side of the story. Please capture everybody.* *You should ask probably commercial fishermen in O'ahu. And Maui, who have more interaction with sharks than we do on [Hawai'i] Island.*

[a] *To keep asking questions… often used in pejorative sense, as of a busybody asking things that do not concern him" (Nīele, 2020). Direct quotes in italics.*

Collaborative Research

This case study and HCTP took collaborative research approaches to enlist fishers in data collection. Researchers employed various strategies to increase fisher participation, from engagement processes (see Methods) to financial rewards for tag deployment. Mutually beneficial outcomes like improved data collection or shark deterrents also incentivized fisher engagement. Fishers described being motivated to participate by their interest in the development of a shark deterrent or handling alternative, and shark behavior and habitat use: "That kind of information might be useful. Then certain times of the year maybe [they're] not around, and the fish are biting, that's when you go… That's another tool in our tool bag when we go fishing." One fisher also contacted the HCTP PI offering to tag sharks for free if funds were scarce, and thanked her for including fishers in HCTP's collection of "real true data."

Key actors, respected and in communication with others, were instrumental in bolstering fisher participation in collaborative research efforts. Interviewees highlighted researchers' and managers' opportunities to use social structures within the community to build trust and share information. One fisher noted key actors' roles in gaining community support: "If you can somehow get the support of the iconic guys… Then they'll spread the word, ah?… Rather than the scientists coming over telling, 'You guys should be doing this'… It's good to garner some support in-house." Several fishers also noted that their decision to participate resulted from friends' encouragement rather than, for example, learning of these projects through printed materials: "I had actually heard about it through the flyers at Pacific Rim, and saw it at the bulletin board and stuff. I just wasn't necessarily gonna go until I heard that other guys were going."

Growth in HCTP participation was observable through attendance at its public tagging workshops. HCTP's first workshop in 2017 was attended by 6 ocean users, most of them fishers. Thirty people attended its second tagging workshop a year later. Afterward, one fisher reached out to congratulate the HCTP team, commenting, "It's typically hard to get that many fishermen to meet for anything. One of the things I got from what you said last Saturday was that getting this kind of participation was a main point of your interest. If that's true, you succeeded." When the lead author returned to Kailua-Kona in February 2019 to share interview results with research participants at a public HCTP workshop, about 30 ocean users attended, including 12 interviewees.

Two-way communication during interviews and HCTP public workshops facilitated trust-building between fishers and

researchers. During these events, fishers asked questions and voiced concerns about researchers' motives and goals, allowing researchers to recognize and respond to them explicitly. This process built trust with participating fishers and encouraged their continued support in data collection and interviewee referral. Interviewees also noted both situational and more general shifts in their own shark-handling practices, including the release of certain shark species, following case study interviews and conversations with the HCTP PI. One fisher, for example, described his friend's remarks upon encountering an oceanic whitetip shark: "[He] told me, 'Ho, I'd kill him but then I thought about [lead author]. I thought, ahh, no.'" Importantly, the lead author took care during interviews not to convey judgment or make arguments against fishers' handling practices. Thus, fisher-researcher communication and relationship-building may also play an inadvertent role in shaping fisher perspectives and behaviors.

In addition to building relationships, trust, and encouraging participation, the HCTP's collaborative research efforts enabled the collection of valuable data that might not otherwise be available. The HCTP trained and equipped fishers for opportunistic shark-tagging. In the 2 years after its first public workshop in 2017, participating fishers deployed 37 tags on oceanic whitetip, thresher, blue, and silky sharks. Prior to this, HCTP researchers independently deployed 15 tags in a comparable 2-year period beginning in 2015, at greater expense in time and finances to the program. Fishers shared their shark-handling practices with the research team in an effort to brainstorm and develop non-lethal shark-handling practices. Some participants were also given special tags to monitor sharks' survival following interaction and, in some cases, jugging. These data expand HCTP researchers' understanding of the impact of small boat fishers' handling practices to sharks.

Knowledge Exchange

Fisher-researcher knowledge exchange was facilitated by collaborative approaches to research, but represents a distinct process in its validation of multiple types and sources of knowledge. Collaborative research and knowledge exchange can, but do not necessarily co-occur. In our case study and the HCTP, researchers and fishers engaged in on-the-water fieldwork and workshop discussions that allowed them to share their perspectives, and demonstrated a willingness to learn from one another. This resulted in shifts of fisher and researcher perspective and, in some cases, behavior. One participant said:

> For once someone's actually going out there with commercial fishermen. Not just one… with multiple. You guys are kinda seeing everyone's point of view. And at the same time, getting everyone to change a little bit toward what you guys see.

One fisher noted the potential collective benefits of fisher, researcher, and manager exchange:

> I think if you create an opportunity that's non-threatening that has nothing to do with taking away their rights, the science and the managers are gonna get a lot of valuable information that they might not otherwise hear, and the fishermen that come… their knowledge and understanding of these species that are important are gonna be dramatically increased.

In the HCTP, both fishers and researchers benefitted from the exchange of knowledge, whether acquired through decades of fishing experience or scientific research. One fisher provided examples from his conversations with the HCTP PI, in which he contextualized tagged sharks' movement data using his knowledge of buoy locations, enlightening her to some sharks' repeated visits to a specific offshore buoy. He noted, "It was really neat though, sharing your knowledge. Like I pointed out to [HCTP PI] about the buoy thing, and she pointed out to me about all the things that I wasn't aware of." This fisher also highlighted an important difference between the ways researchers and fishers like himself perceive the fisher-shark interaction problem:

> Most of the scientists feel that we… don't like the sharks "cause" they eat [our] fish. I can count the number of times on one hand that a shark's actually attacked my fish. The problem is when they're in the area, we can't catch fish. They create a barrier.

Knowledge exchange was not a primary goal of the HCTP. But, as a result of participants' respect for both experiential and scientific knowledge, it improved fishers' and researchers' understanding of shark interactions.

DISCUSSION

This case study took advantage of managers' and researchers' growing interest in the oceanic whitetip shark to explore the poorly documented fisher-shark interactions of Hawai'i. Stepping back from the dominant shark conservation narrative, we engaged West Hawai'i small boat fishers to understand from their perspective what problems frame fisher-shark interactions. This early investment in problem diagnosis is critical to develop solutions that are not only effective, but avoid intensifying any existing conflicts (Zimmermann et al., 2020). We used our adapted framework (**Figure 1**) to examine the layered conflicts defining this "problem" beyond threats to pelagic sharks, and reflect on potential solutions better oriented to coexistence. Prominent among dispute-level problems was decreased fishing efficiency. We documented some applications for substantive solutions like information provision, financial incentives, and the development of shark-handling alternatives. However, diversity among fishers and the various factors interviewees described guiding their behavior on the water necessitate integrated, multi-pronged solutions that also account for deeper conflicts. Deeper conflicts revealed in this case study were contextualized by relationships, power dynamics, and identity-based conflicts.

Interviewee definitions of the fisher-shark interaction problem were dominated by dispute, referencing sharks as fishing competitors, financial costs of an interaction, and fishers' decision-making factors during shark encounters. However, our findings reinforce that the meaning fishers ascribe to sharks (**Figure 3**) varies according to species and context (Molony and Thomson, 2020), with emotional, cultural, safety, and

economic considerations (Glaus et al., 2019). Upon encountering a shark, fishers included situational shark attributes, landing opportunity, physical capacity, time and finances invested, and social pressures among their decision-making variables (**Table 2**). These represent a number of individual capacity, economic, and social norms variables that drive fisher behavior and merit further investigation (St. John et al., 2010). To the extent that these variables predict fisher behavior, interventions may enable alternative decision-making pathways (Fulton et al., 2011); for example, by mitigating fishers' financial costs.

Interviewees offered some substantive solutions that address these goals, including providing information, compensating fishers, and developing shark-handling alternatives. Some interviewees described shifts in their perspective or behavior after building relationships with researchers or learning about sharks' biology or threatened status. However, the diversity of West Hawai'i fishers' demographics, values, attitudes, and capacity for behavior change around sharks necessitates more than social influence and information provision (Stern, 2000; Campbell and Cornwell, 2008; Reddy et al., 2017). Other solutions may help to address this diversity. Our results suggest that financial incentives may promote fisher engagement and increase fisher access to a broader suite of behavioral and shark-handling options, including shark-tagging. Financial incentives may benefit fisheries management goals and cost-effectiveness (Innes et al., 2015), but their success is conditional (Bladon et al., 2016). Fisheries bycatch mitigation may benefit from the application of financial incentives with multi-pronged approaches that also prioritize, for example, collaborative research or efforts to support community leadership and stewardship capacities (Lent and Squires, 2017; Milner-Gulland et al., 2018). Interviewees also expressed interest in a shark-handling alternative or deterrent, making suggestions for non-lethal handling strategies that might preserve fishing opportunity. This illustrates the collaborative space that exists for fishers, researchers, and managers to pursue solutions with collective benefits for fishers, sharks, and those invested in shark conservation.

Although we documented success in applying substantive solutions to disputes, we also recognize that solutions may cross-cut levels of our framework. The interpretation that levels of conflict require solutions at equal and potentially shallower levels (Zimmermann et al., 2020) may obscure innovative problem-solving. For example, fishers' perception of sharks as threats to fishing opportunity has been connected to their diminished support for shark conservation (Drymon and Scyphers, 2017). However, fishers' participation in this study and the HCTP highlights opportunities to dissociate the two. Firstly, like U.S. recreational fishers who expressed a lack of concern for "nuisance" sharks frequently caught as bycatch (McClellan Press et al., 2016), interviewees devoted little attention to sharks historically, characterizing them as incidental, non-target species. Secondly, interviewees described their increasing attention to sharks–and in certain cases, adopting less harmful handling practices following fisher-researcher engagement–over the course of this study. Along with the role of social pressure in determining fishers' behavior, these examples underline the potential for social norms (Nyborg et al., 2016), influential actors (St. John et al., 2010), and fisher-researcher relationships (Campbell and Cornwell, 2008) to affect fishers' participation and on-the-water behavior. If researchers and managers seek only substantive solutions to disputes, they may miss opportunities to change fisher attitudes and behaviors through process- and relationships-based approaches.

Substantive solutions may also generate deeper conflicts instead of resolving disputes as intended. Given the deeper conflicts described by interviewees and the region's absence of enforcement (Tissot et al., 2009), regulation may not only fail to achieve its goals but critically impede other fisheries management efforts. Following regulation, fishers may assign additional negative meaning to sharks, further complicating sustainable resolution of fisher-shark interactions as has been seen with other species (Nie, 2001; Naughton-Treves and Treves, 2005). Inappropriate substantive solutions may also exacerbate stakeholder conflict (Redpath et al., 2013) and diminish fishers' perceived legitimacy of management and science. Degraded legitimacy is of concern given the benefits of legitimacy for compliance (Levi et al., 2009), particularly when enforcement is lacking (McClanahan et al., 2006) as in the case in West Hawai'i. Deeper conflicts like these may then obstruct parallel and future efforts to mitigate fisheries management problems involving the same actor groups, whether or not the dispute is focused on sharks. In this case study, for example, researchers encountered and addressed feelings of apprehension from past fisher-researcher interactions (**Table 4**) to build trust and improve collaborative research efforts. Thus, degraded relationships, mistrust, and unresolved conflict at once challenge problem-solving endeavors (Schuckman, 2001; Ansell and Gash, 2007) and highlight opportunities for creative, collaborative solutions.

Awareness of deeper conflicts is therefore critical to effective problem-solving. This study illuminated issues of power, historical relationships, and identity in fishers' problem definitions that necessitate relationships- and process-based solutions (Webber et al., 2007; Campbell and Cornwell, 2008; Penney et al., 2017; Shiffman et al., 2017; Crespin and Simonetti, 2019). Interviewees described concerns about fisheries management and science that degraded fishers' perceptions of their legitimacy (**Table 3**). Among them, disconnect between fishers' and researchers' or managers' knowledge and vulnerability to fisheries management, politicized science and decision-making, poor enforcement, data validity, and transparency. Interviewees also perceived certain actors, such as fishing industry or environmental groups, to have greater organizational capacity, financial capital, and access to management decisions—forms of power that play important roles in conservation contexts (Schuckman, 2001; Chapin, 2004). When regulations were perceived to be inequitable or politically motivated, interviewees doubted management logic and efficacy. Despite fisheries scientists' and managers' growing recognition of the value of stakeholder knowledge (Neis et al., 1999; Reed et al., 2007; Wendt and Starr, 2009), interviewees also noted power differentials resulting from fishers' lack of formalized knowledge or specific language through which input is typically valued. Designing engagement to increase access and participation for groups that possess alternative forms of knowledge, capital,

or language can help to address these imbalances. Fishers also perceived management to deny their agency, connection to place, and stewardship. Solutions that ignore issues of identity may fail to resolve conflicts as intended (Hicks, 2001; Rothman and Olson, 2011; Doucey, 2011). Finally, interviewees expressed concerns about being misrepresented or contributing to a process that would result in fishing closures or restrictions (Silver and Campbell, 2005; St. Martin and Hall-Arber, 2008; Carruthers and Neis, 2011). To improve outcomes, fisheries solutions might deliberately consider fisher identities, cultures, and physical and financial risks (Faasen and Watts, 2007; Coulthard et al., 2011; Rivera et al., 2017).

To address layered problems and avoid generating new, deeper conflicts through singular approaches, robust fisheries management solutions should consider integrated, multi-pronged approaches (Madden and McQuinn, 2014; Ayers and Leong, 2020; Booth et al., 2020). To this end, our work demonstrates potential for collaborative research and knowledge exchange to incorporate substance-, process-, and relationships-based elements. Through regular communication, researchers shared information about shark biology and management statuses, outcomes of collaborative research, and responded to fishers' concerns about researchers' motives and goals. Engagement included early disclosure of possible risks and outcomes of engagement, which helped to establish fisher-researcher trust and reciprocity. Key actors and face-to-face interactions encouraged information sharing and certain behaviors (Abrahamse and Steg, 2013; Mbaru and Barnes, 2017), including shark release. Key actors were also instrumental in recruiting participants and gathering data. As engagement exerts high, prohibitive transaction costs on its participants, requiring investments in time, energy, and money (Ayers et al., 2017), researchers made efforts to minimize these costs to better include diverse fisher perspectives. Other key process-based engagement elements include foundations in trust, equity, and learning, explicit goals (Reed, 2008), facilitative capacity, and transparent decision-making (Reed, 2008; Vaughan and Caldwell, 2015; Mease et al., 2018).

Collaborative research efforts documented in this case study improved fishers' and researchers' understanding of fisher-shark interactions and enabled the collection of otherwise inaccessible shark interaction data through interviews and tagging. A collaborative approach may improve the cost-efficiency of data collection (Mackinson and Nøttestad, 1998). Just as local, participatory processes may improve the perceived justness and legitimacy of regulations (McClanahan et al., 2006), collaborative research may improve the credibility of resulting data for both researchers and fishers (Hartley and Robertson, 2006; Wendt and Starr, 2009).

Collaborative research and knowledge exchange also forged a shared space to pursue novel on-the-water strategies to mitigate fisher-shark interactions. As fishers and researchers exchanged knowledge, both groups acquired new understanding of the problem by recognizing the value in both experiential and scientific knowledge (Hartley and Robertson, 2006). This learning process is crucial to bridging stakeholder disconnect and facilitating productive discourse (Adams et al., 2003). Engagement processes that facilitate the sharing and adaptation of conflicting values and interests are likely to benefit outcomes' legitimacy and durability (Søreng, 2006; Redpath et al., 2013).

CONCLUSION

Complex fisheries problems are often identified and managed based on biophysical, dispute-level problem definitions. This study delivers a foundational inventory of localized sociopolitical context relevant to pelagic sharks and their interactions with West Hawai'i small boat fishers. By identifying factors that shape fishers' diverse perceptions and behavior around sharks, researchers, and managers, we take a crucial first step toward strategies that are more cost-effective, equitable, and prepared to achieve their goals (Booth et al., 2019). While others have applied Madden and McQuinn (2014) framework to examine conflict in various contexts, including human-wildlife conflict, we adapted it to uncover hidden problems and solutions. This allows framework users to question monolithic problem definitions, account for stakeholder politics and histories in problem-solving which may appear unrelated to the dispute, and identify potentially harmful management pathways.

The fisher-researcher exchanges integral to this study and the Hawai'i Community Tagging Program improved fisher access to management discourse, fisher-researcher relationships, and both groups' understanding of sharks and their interactions with Hawai'i small boat fisheries. These outcomes indicate that collaborative solution-finding efforts could reduce costs and negative outcomes and produce novel benefits to local fishers, sharks, and the research and management communities invested in them.

Pelagic sharks represent an international policy concern given their mobility and global distribution. This case study illustrates that the sociopolitical contexts for pelagic shark management not only require diagnosis at local scales, but reveal process- and relationship-based solutions that international policy may not be able to provide. This framework can be applied locally to illuminate deeper conflicts and innovative solutions wherever humans and wildlife might coexist. Furthermore, the framework has utility wherever we seek solutions amidst dissonant problem definitions, threatened identities, fractured relationships, and power inequities.

DATA AVAILABILITY STATEMENT

The datasets presented in this article are not readily available due to confidentiality considerations, data can only be shared anonymously, in aggregate. Requests to access the datasets should be directed to Mia A. Iwane, mia.iwane@noaa.gov.

ETHICS STATEMENT

This study was reviewed and approved by University of Hawai'i Institutional Review Board. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

MI collected and analyzed data, adapted the theoretical framework to synthesize results, and wrote the manuscript under the guidance of her thesis committee members KO, KL, and MV. KO served as the committee chair, providing oversight along with KL throughout the project. All authors provided critical feedback to shape the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.669105/full#supplementary-material

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Transparency About Values and Assertions of Fact in Natural Resource Management

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Worldwide, unsustainable use of nature threatens many ecosystems and the services they provide for a broad diversity of life, including humans. Yet, governments commonly claim that the best available science supports their policies governing extraction of natural resources. We confront this apparent paradox by assessing the complexity of the intersections among value judgments, fact claims, and scientifically verified facts. Science can only describe how nature works and predict the likely outcomes of our actions, whereas values influence which actions or objectives society ought to pursue. In the context of natural resource management, particularly of fisheries and wildlife, governments typically set population targets or use quotas. Although these are fundamentally value judgments about how much of a resource a group of people can extract, quotas are often justified as numerical guidance derived from abstracted, mathematical, or theoretical models of extraction. We confront such justifications by examining failures in transparency about value judgments, which may accompany unsupported assertions articulated as factual claims. We illustrate this with two examples. Our first case concerns protection and human use of habitats harboring the northern spotted owl (*Strix occidentalis caurina*), revealing how biologists and policy scholars have argued for divergent roles of scientists within policy debates, and how debates between scientists engaged in policy-relevant research reveal undisclosed value judgments about communication of science beyond its role as a source of description (observation, measurement, analysis, and inference). Our second case concerns protection and use of endangered gray wolves (*Canis lupus*) and shows how undisclosed value judgments distorted the science behind a government policy. Finally, we draw from the literature of multiple disciplines and wildlife systems to recommend several improvements to the standards of transparency in applied research in natural resource management. These recommendations will help to prevent value-based distortions of science that can result in unsustainable uses and eventual extinctions of populations. We describe methods for communicating about values that avoid commingling factual claims and discuss

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approaches to communicating science that do not perpetuate the misconception that science alone can dictate policy without consideration of values. Our remedies can improve transparency in both expert and public debate about preserving and using natural resources, and thereby help prevent non-human population declines worldwide.

Keywords: policy, preservation, owl, sustainable use, wolf, model, research conduct, scientific integrity wolf

INTRODUCTION

Worldwide, unsustainable use of nature threatens the collapse of ecosystems and the benefits they provide to non-humans and humans alike (Ceballos et al., 2015, 2020; Darimont et al., 2015; Ripple et al., 2017; Ceballos and Ehrlich, 2018; FAO, 2020). Yet, for at least a century, governments and researchers have invoked a long-standing set of scientific models as a basis for the claim that natural resource extraction (particularly of wild animals) is sustainable (Larkin, 1977; Oro, 2013). How can a well-understood and established science of "sustainability" commonly result in unsustainable extraction? We attempt to resolve the seeming paradox by describing the need to disentangle where the science begins and ends in natural resource management (NRM), and how uncertainty in science and mistrust of scientists are both exacerbated by a lack of transparency about value judgments (judgments about what one considers desirable or undesirable for goals and modes of conduct; Manfredo and Dayer, 2004). We acknowledge that all of us struggle to demarcate our observations and inferences about the world from what we desire to be true about the world. Yet, scientists, in particular, should strive for such clarity and openness because the ideal of objectivity in science means that our desires should be regarded as a poor guide to our approximations of reality.

We begin with disentangling where the science begins and ends in NRM use and policy. Governments often set targets (e.g., population goals or use quotas) for the abundance of nature's components that people want to use. Decisions on targets, goals, or quotas (i.e., annual use of populations) necessarily involve value judgments external to the science for answering how much to use or preserve: Should we extract a given component of nature? How? Where? And—importantly—how much should we use now or preserve for others and the future? Population goals and extraction quotas set by governments represent choices of certain quantitative amounts among many possible amounts. That choice is based in part on judgment and preference about how much humans ought to need or ought to take, not on mathematical models. We define science as observation, measurement, analysis, and inference following long-accepted principles of transparency, reproducibility, and impartiality. Science can only tell us how much to use or preserve to achieve a predetermined goal as guided by values.

Entanglements of science and values arise when people improperly conflate personal or community preferences with evidence at hand when justifying their decisions (Schrader-Frechette and McCoy, 1994; Menon and Lavigne, 2006; Artelle et al., 2014; Darimont et al., 2017, 2018; Artelle K. A. et al., 2018; Artelle K. et al., 2018; Jacquet and Delon, 2018). Science can only describe how nature works and predict the likely outcomes of our actions, or sometimes reveal the possible range of actions, as discussed for biodiversity and natural resources (Lynn, 2006, 2010; Nelson and Vucetich, 2009; Treves et al., 2009b; Vucetich and Nelson, 2014; Artelle K. et al., 2018; Jacquet and Delon, 2018). Value judgments can be concealed by suggestions that scientific theories or mathematical models command a limited range or single choice, such as setting a goal or quota.

Environmental sciences underlying policy and management of fisheries, wildlife, biodiversity, climate, etc., risk tangling personal, institutional, or societal preferences or value judgments with quantitative values derived from extraction models. For example, deciding how many fish to extract is informed by, but not the same as, inferring how non-human consumers might respond to the resulting population of fish, as the former inevitably involves considerations of the relative value of fish and other wildlife, along with other costs and benefits (Levi et al., 2012). Likewise, the amount of CO_2 emissions represents a measurement of a quantity, but setting the allowable level is a value judgment that entails balancing benefits and costs for health, wellbeing, resources, equity, diversity, ecosystem function, and any other criteria the decision makers choose to integrate.

What role do facts—and assertions of fact (hereafter fact claims)—play in justifying NRM goals? When public policy is being made, interest groups commonly call upon certain facts (or claims thereof) to justify their preferred policy outcomes. Science is valuable to assess the accuracy, precision, and reliability of such putative evidence before we can label it as fact or unverified assertion. Given competing values—and the inherent uncertainty that pervades scientific understanding, scientific processes should therefore keep evidence (and the processes generating it) transparent and accountable for the public, decision makers, and regulators. Transparency should also include a clear acknowledgment about the value judgments before asking scientific questions, such as "how much is out there to use; how much can we use without harming competing claims that we or others also value; and how much are we leaving for other users of the present and future?" Clearly, there is an interplay among values, fact claims, and the evidence (scientific inference about the uncertainty of the fact claims).

Scientists seem to need ways to keep claims about both facts and values in public affairs transparent and accountable. Values often play a dominant role in policy setting (Schrader-Frechette and McCoy, 1994; Lynn, 2006; Nelson et al., 2011; Darimont et al., 2017; Artelle K. et al., 2018). Scientifically inclined societies have developed robust (if fallible) systems for vetting fact claims through many processes and agents, such as

scientific peer review, replication of research, inspector generals, and third-party watchdogs.

Our concern here lies at the intersection of science and values; how scientists and policy-makers make transparent their value judgments, separate these from fact-claims, and evaluate their own claims and those of others transparently. Our goal builds on a call by the National Academies of Science, Engineering and Medicine (2017) in which they clearly articulated the nature of our concern, offering, "Openness is fundamental to the success of the entire chain of processes and relationships involved in scholarly communication. Most centrally, those assessing the quality of science must be honest in their assessments and aware of and honest in reporting their own conflicts of interest or any cognitive biases that may skew their judgment in self-serving ways…" p. 33 (National Academies of Science, Engineering and Medicine, 2017). The arena in which such biases require addressing is in communications, where the tangles of values, facts, and fact claims are written or spoken to justify action or inaction. Often scientists are required to extrapolate beyond the science into more speculative ground. It is precisely in communications of scientific observations, measurements, and inferences that one finds extrapolation and uncertainty raise the risk of false fact claims, and transparency may be low. Therefore, our first case study examines the debate among scientists and between policy scholars and scientists regarding how verified, scientific evidence ("facts" hereafter) should be communicated in a policy debate founded on value judgments, in which the science will lead directly to action or inaction toward owls or their habitats (**Box 1**).

In the spotted owl case study, we reason that transparency on the parts of all scientists commenting on public policy would go a long way to revealing in the broad public when a value judgment has been made about a policy goal, when a fact claim has been made to justify action toward that goal, and when a fact claim is being evaluated scientifically. Training for such transparency has been taken up under the rubric of research integrity or scientific ethics, and we welcome the recent spread of such efforts (Mejlgaard et al., 2020). Our next case study raises questions about the internal domain of the conduct of science (observation, measurement, and inference) and specifically how sustainable use models and outputs might conceal value judgments. Before exploring the next case, we summarize the science of sustainable use models widespread in NRM.

TRANSPARENCY IN SUSTAINABLE USE SCIENCE

We focus on transparency because its pursuit and its breach illustrate how value judgments from the external domain can infiltrate and distort the conduct of science (its internal domain). Transparency can help an agency avoid overwhelming bias or help watchdogs check such biases before they do harm. Our concern with transparency begins with the wording used in our topic: sustainable use of natural resources. The phrase "natural resource" itself presupposes a value judgment about human claims to nature. Indeed, the field is littered with open or buried value judgments and euphemisms (Houck, 2001; Mark, 2014; Johns and DellaSala, 2017). For example, the word "exploitation" often surfaces in NRM. The OED (2020) defines "exploit" first as, "To harvest or extract (a natural resource); to extract resources from (a place)," but its second definition is, "To take advantage of in an unfair or unethical manner; to utilize for one's own ends." Given that exploit potentially enfolds a value judgment as in the latter definition, we encourage replacement with the simpler terms "use" and "extract," as in the former definition because "extract" does not enfold a value judgment: "Taken out, obtained out of something… Derived (from a source)" (OED, 2021). Many other euphemisms surface in NRM, e.g., "take," "harvest." Euphemisms and jargon pose many problems (Johns and DellaSala, 2017), but even those who use neutral terminology may overlook value judgments within mathematical models for the use of nature, as we examine next.

Estimations of the point of maximum sustainable yield (MSY) and forecasts of population change after extraction have dominated the science of NRM for decades (Larkin, 1977; Nelson et al., 2011; Levi et al., 2012; Patrick and Cope, 2014). Predictions of sustainability for points on a hypothetical curve (the science) can be confused with a value judgment.

The value judgment often used is that extractors should aim for that rate of extraction for current human consumers, and the extraction should be repeated to achieve sustainable preservation of the resource for future humans. Conflating these different components of decisions about use can mislead anyone into missing the associated personal or organizational value judgments (How much should we use and how much should we preserve for others? Who are the others, and what do we owe them?). We expose these issues to a clearer light for NRM by juxtaposing the mathematics against the value judgments that often follow such modeling.

Prevalent models of the sustainable use of natural resources often apply a theory of population dynamics called density dependence. Density-dependent dynamics arise when reproduction changes non-linearly with population density, or when mortality changes non-linearly with population density. A member of the family of logistic growth curves can describe such population dynamics. This family of curves has a convenient mathematical form that includes an inflection point between asymptotes at zero (local extinction) and carrying capacity (K) for a hypothetical closed ecosystem (**Figure 1**). The theory suggests that the use of natural resources at many points along the curve would be sustainable because new births could replace deaths caused by users. The point of maximum sustainable yield (MSY) is predicted to occur at the inflection point of the logistic growth curve in its simplest form, usually assumed to fall at half of K (**Figure 1**). The many uncertainties about the theory are well known (**Figure 1** caption). Even today, when MSY usually is not chosen as an explicit goal, modelers and managers often select a "safer" extraction point higher up the curve of presumed population growth (Kirkwood, 1981), still with implied reference to MSY. Even though some modelers have long recognized the risks of setting population goals, targets, or quotas, they have often only recognized the uncertainties about MSY but not the problem of buried value judgments about who benefits and who loses from such extraction. All the

BOX 1 | Debates over spotted owls reveal how scientists grapple with transparent value judgments in scientific communications.

Northern spotted owl (*Strix occidentalis caurina*) policy in the USA has been subject to repeated debate about biased scientific communications and embedded value judgments influenced by laws (Carroll et al., 2012; Wilhere, 2012; Wilhere et al., 2012; Peery et al., 2019; Rohlf, 2019). Simplifying the 2012 debate somewhat, Wilhere et al. (2012) criticized the scientific peer reviewers contracted by the US government under the Endangered Species Act (ESA) for not transparently explaining where the science began and ended. Wilhere et al. suggested that the scientists did not separate facts, whether measurements of the current status or inferences about future conditions of the owls under hypothetical policy scenarios, from the value judgments that must underlie recommendations to the US government. Again simplifying, Carroll et al. (2012) countered that an ESA peer review is embedded in a set of statutory value judgments about listing and delisting owls. The debate might have been just another example of one team of scholars citing another for bias and being rebutted, if not for the following elements of their debate that hold broad general interest.

Wilhere (2012) suggested that the scientists in the peer review panel had prioritized protection of the owls rather than treating the two policy options (protect or not) impartially. He went on to recommend that the government "…separate discussions of policy from discussions of science. In fact, consider submitting separate policy and science reviews authored by different groups and published as separate reports" p. 747 (Wilhere, 2012). Later, the US government tried something similar with a separate ethics review (Lynn, 2018; Andrews et al., 2019), but, to our knowledge, did not publish the recommendations. We are not aware if anyone followed another recommendation by Wilhere (2012) that policy-makers purge policy advocacy from scientific reviews (see a deep look at this issue in Doremus, 2004), though he neither recommended the converse nor examined the possible shortcomings of such a process.

We surmise it is no easier for scientists to articulate all their value judgments than for policy-makers to resist making unverified fact claims to support their value judgments that underlie policy proposals. Are we left with no resolution to this confounding interplay of facts and values? The next half of our case study of spotted owl science communication reveals an important dividing line between values about how science is conducted (the internal domain of research integrity) and values about how science is communicated to outside audiences (the external domain).

Peery et al. (2019) denounced another owl research team's verbal and legal actions against unnamed junior scientists. We echo their call for civil debate and for presenting facts in peer-reviewed journals that follow ethical guidelines. However, they also demarcated acceptable and unacceptable modes of communication and styles of verbal expression by scientists communicating science. Their **Supplementary Table 1** (Peery et al., 2019) shows the "symptoms" of agenda-driven science that intermingled personal value judgments about style, tone, and modality of communication with concerns about bias, with concerns how science should be conducted. We acknowledge that a tabular format may force authors into shorthand. Examples of this intermingling include, when labeling "Activities symptomatic of agenda-driven science," they included "Failure to disclose involvement in litigation related to a study." They probably intended something akin to competing interests in litigation over a study, but the recommendations will be quoted by others, so we address them as written in **Supplementary Table 1** (Peery et al., 2019). Our aim is to make clear that value judgments and debate about scientific communications should distinguish the internal domain (how the science was conducted) from the external domain of science communication (how the science was disseminated) because transparency about those two domains will help demarcate where value judgments concern personal or organizational differences of opinion and where judgments concern the validity of facts and fact claims.

We begin with the series of "Activities symptomatic of agenda-driven science" that Peery et al. use to address their concerns over how spotted owl science was conducted (the internal domain). For example, they cited: "Selective use of data…," "Selectively referencing…," "Emphasizing certainty and simplicity," and "Conducting biased review…" (**Supplementary Table 1**; Peery et al., 2019). These are poor practices in the conduct of science (National Academies of Science, Engineering and Medicine, 2017) but are also practices that might be construed as sloppy if unintentional. Although Peery et al. (2019) extensively documented their evidence for intention by one team of owl researchers, it is less clear if these practices identified in **Supplementary Table 1** are truly "agenda driven" in all cases.

Indeed, if the scientific community were held to some of the standards in **Supplementary Table 1** of Peery et al. (2019), almost everyone would be judged as "agenda driven." Poor practices lie alongside practices that might actually be good practices in their **Supplementary Table 1**. For example, depending on how one defines pressure, "Pressuring other scientists to retract…" might be good for research integrity, given that the Committee on Publication Ethics (COPE, 2019) recommends how retraction should proceed when third parties ask editors or publishers to investigate cases of possible research misconduct. Likewise, one of the proscriptions of Peery et al. (2019) seems to suggest that once a paper passes peer review, it is immune to criticism thereafter, in the phrase "…outside of the peer review process" **Supplementary Table 1** (Peery et al., 2019). Given that post-publication review is now being recommended by prominent journals and ethicists for improving reproducibility of science (Allison et al., 2016; Stern and O'Shea, 2019), the proscription on timing of criticism is yet another value judgment by Peery et al. (2019).

Although the last two value judgments might be excused as traditional facets of scientific integrity giving way to new norms, other value judgments in **Supplementary Table 1** of Peery et al. favor some actors over others within scientific debates rather than favoring some methods or evidence over other methods or evidence. If agendas should not drive science, then personal, organizational, or governmental preferences should not either. We have considerable concern for science driven by preference. Basing scientific inference or evaluation of the quality of science on the identities of the authors, their funders, or their access to data seems bad for efforts to observe and infer reality, which should be judged on their merits alone (National Academies of Science, Engineering and Medicine, 2017). Two of the recommendations in **Supplementary Table 1** (Peery et al., 2019) strike us as containing hidden value judgments about the establishment or status quo being good and others as bad. For example, the phrases "…unvetted data" and "…through [Freedom of Information Act requests (FOIA)]" treat established, traditional, government-funded or government-run research as good. Yet, litigation or FOIA requests can be an effective recourse to obtain information from secretive agencies, programs, or individuals (see Rohlf, 2019 for advocacy of scientific engagement in litigation and Doremus, 2004 for a history of non-transparency by agencies implementing the ESA). Similarly, litigation, social media, and critique of scientific papers in public media, comprise legitimate means of communication in a free, democratic society. None of these actions in the external domain of science (how it is communicated) are intrinsically bad for the internal domain of research integrity (Lackey, 2007; Garrard et al., 2016; National Academies of Science, Engineering and Medicine, 2017). By asserting norms of scientific communication, Peery et al. (2019) appear to have advocated for their values related to decorum, style, or modality. We doubt that public confidence in science will be increased by censorship of scientists. We fear a scientific culture that sanctions legitimate science communication is the next step if the "symptoms" in **Supplementary Table 1** (Peery et al., 2019) were widely sanctioned in norms of scientific communication.

mathematical and probabilistic reasoning about uncertainty is additive to the point about the external domain of science—choosing specific population targets or quotas involve value judgments independent of uncertainty in the model outputs.

MSY is just one point among an infinite number of points on an uncertain curve. The decision to set a population goal at MSY or any other point is just one alternative judgment about many possible values. Those are decisions that usually prioritize

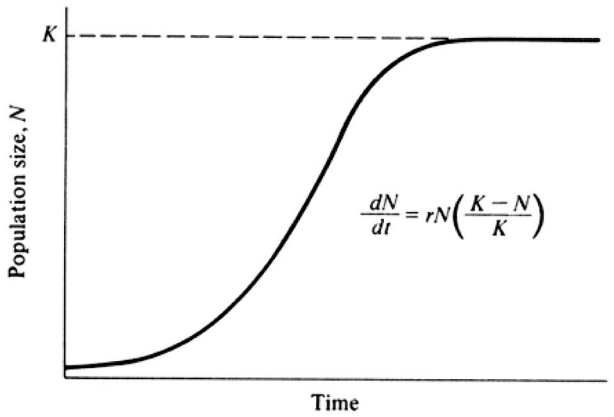


FIGURE 1 | The logistic growth curve resulting from density-dependent population dynamics. *K* is the asymptote or predicted carrying capacity of an idealized population and *r* is the intrinsic rate of increase. The equation is positioned at approximately the same height as the inflection point K/2, which is predicted to be the point of maximum sustainable yield, MSY (Pianka, 2000). Defined simply, the carrying capacity (K) is the maximum number of individuals that can be supported in each area at a given time. In populations following density-dependent population dynamics, the upper asymptote of the logistic growth curve is assumed to approach K because the average death rate and rate of recruitment of juveniles to adulthood equilibrate, and growth stops. The presence of carrying capacity, the prediction of logistic growth, and the point of MSY are all testable predictions of a general theory that may not be accurate *or even exist* in specific cases. Indeed, real populations vary in the shape of their growth curves, and these characteristics can change over time (Brook and Bradshaw, 2006; Ratikainen et al., 2008). Birth and death cannot be assumed to change symmetrically or non-linearly with density or population size. For example, with strictly territorial species, local density may not even increase much within a local subpopulation (see **Box 2** for an example). Even if the other conditions are met, the inflection point can lie higher or lower than K/2, depending on the symmetry of the birth and death curves in relation to density. Caution is warranted because of the vicissitudes of wild ecosystems, environmental stochasticity, and the uncertainty of any predictions about the future that are based on past patterns (Brook and Bradshaw, 2006; Ratikainen et al., 2008; Oro, 2013; Williams, 2013). Data limitations especially relating to migration into and out of the open ecosystems also commonly restrict our knowledge of growth rates of natural populations. Also, the predictions of sustainable use models become increasingly uncertain when mortality and natality are affected by unregulated factors, untested policies, or human errors (Fryxell et al., 2010). This is not to say that MSY is a fallacy, but rather that it is a prediction that needs frequent updating, continual review, and testing, rather than an *a priori* static point.

certain human users (i.e., current and often influential interest groups). Indeed, a rigid application of MSY benefits current human extractive users at the expense of all other users, often including non-consumptive users or non-human consumers who also use prey populations. That contrasts with other definitions of sustainability, in which, for example, unused individuals are not viewed as "wasted," but instead as feeding other life, preserved for the future, or serving other ecosystem functions, e.g., salmon *Oncorhynchus* spp. fisheries (Levi et al., 2012). Also, the interests of future generations are often presumed (rather than consulted explicitly or preserved for all possible uses) when current decision makers allocate to current human users (Treves et al., 2017b, 2018a). Finally, the very conception of nature as a resource that is claimed property privileges an anthropocentric, western worldview to the detriment of other lifeways, especially and commonly those of indigenous peoples and non-humans (Lynn, 2006; David, 2009; Levi et al., 2012; Artelle K. et al., 2018; Eichler and Baumeister, 2018; Treves, 2019a). Decisions to allocate natural resources often blur the line between real need for subsistence or survival and mere profit seeking, recreation, or luxury (Santiago-Avila et al., 2018b; Santiago-Ávila et al., 2020; Treves et al., 2018b; Treves, 2020). MSY thinking and density-dependent models of population dynamics applied to NRM illustrate the internal and external domains of science and their relationship to values well.

The judgment whether one should or should not extract a given quota stems from values, which may be followed by a judgment whether to apply a density-dependent model, itself stems from scientific reasoning. Later, the judgment whether to act on the model predictions again stems from values informed by the consequences that science estimates with uncertainty. That three-step process involves interaction between values and science. Therefore, transparency about how much will be extracted, how and where to limit non-human populations, and which actors are expected to benefit or suffer, would expose the value judgments and outcomes in a policy decision or management action associated with a population target or extraction quota. We illustrate the consequences of non-transparent value judgments further complicated by non-transparent conduct of science with a case of gray wolves (*Canis lupus*) (**Box 2**).

GENERAL CONCLUSIONS FROM THE CASE STUDIES

Our cases reveal two features of transparency in science and public policy. The first case about northern spotted owls explained how scientists and scholars in policy-relevant research debated the thoroughness of transparency about value judgments and whether scientific debate could be purged of personal value judgments and how that might occur. We use the spotted owl case to argue for clearer thinking about—and behavior toward—the difference between the internal domain of science (how it is conducted) and its external domain facing the public and policy-makers (how it is communicated). In our second case, a government policy on population targets, quota levels, and wolf hunting was promoted by science that lacked transparency and from which—we infer—the value judgments of a handful of scientists and perhaps their institutions remained undisclosed but might have played a disproportionate role in guiding management for 21 years. The wolf case study began somewhat transparently with explicit statements, such as "The management goal represented the minimum level at which a full array of population control activities could occur including pro-active depredation control and the possibility of public harvest" p. 16 (WDNR, 1999). However, more fundamental questions were never addressed, such as "Should we kill wolves? And if so, why, what are the desired outcomes, under what conditions is it right and effective to pursue those outcomes, and how many wolves is it right to kill to attain the goals?" (Treves, 2009). This

BOX 2 | Science used to justify a gray wolf population goal and kill quota, 1999–2021 now.

Here we explain how a population model used to plan the regulated killing of wolves in Wisconsin, USA, was not transparent about value judgments in conducting or communicating the science marshaled in support of a use policy. The senior author here observed public policy and re-analyzed data from published reports in recent years (Treves, 2019a; Treves et al., 2014, 2017a,b,c) after collaborating with the state wildlife agency (WDNR) previously (Treves et al., 2002, 2004, 2009a, 2011; Wydeven et al., 2004; Olson et al., 2014); see the lead author's full declarations of potentially competing interests here (http://faculty.nelson.wisc.edu/treves/Vision.php accessed 29 October 2020).

The state of Wisconsin published its first wolf management plan in 1999 (WDNR, 1999), which included a projection of wolf population growth to 2020 (**Figure 2**) from the estimated minimum of 205 adult and yearling gray wolves in winter 1998–1999. The 1999 Plan codified two numerical value judgments that shape policy to this day. The first, the Delisting Level of 250, was set higher than the federal ESA delisting level of 100 (USFWS, 2020). The Delisting Level was set at the midpoint of the range of outputs of a population viability analysis (PVA), which the authors argued "needs to be cautiously interpreted and should not be used by itself to set management goals. Based on [the PVA], a population between 200 and 300 seemed appropriate for delisting wolves in Wisconsin" p. 16 (WDNR, 1999). We note the questionable idea that a model output could set management goals without policy-makers stating a goal based on value judgments. The second value chosen by the authors of the 1999 Plan was the Management Goal $N_{goal} = 350$ (**Figure 2**), also referred to as the "population goal" in WDNR, 1999 or "population objective" as of this writing (https://dnr.wisconsin.gov/topic/WildlifeHabitat/wolf/index.html accessed 28 October 2020).

By definition, $N_{goal} = 350$ is a value judgment, but this choice was couched in models and several fact claims as follows: Ostensibly, N_{goal} was a number of wolves supportable by suitable habitat, compatible with the PVA, and was also "socially tolerated" p. 15 (WDNR, 1999), i.e., "… a reasonable compromise between population capacity, minimum level of viability, and public acceptance." p. 16 (WDNR, 1999). As quoted above, the PVA seemed to be regarded ambiguously by the authors and appeared to have little or no influence in policy setting. Regarding "social tolerance" and "public acceptance," the 1999 Plan reported an informal survey about N_{goal} after it had already been selected, with the following statement of methods: "During the review of the second draft of the [1999 Plan], of persons commenting on the population goal, 38% supported the goal, 38% felt it was too low, and 24% felt it was too high…" p. 16 (WDNR, 1999). From the wording, we infer that N_{goal} was presented to the assembled individuals (of which a sample size and selection procedure were not stated). Methods were not disclosed, and independent review was apparently not conducted, as with most agency management plans in North America (Artelle K. A. et al., 2018). Therefore, the assertions about public opinion or social tolerance seems an untested fact claim. The third fact claim about N_{goal} was about "population capacity" or "suitable habitat" p. 15–16 (WDNR, 1999). Two estimates existed for carrying capacity, K, in 1999. One was based on habitat at K = 300–500 wolves (or possibly up to 800 if marginal habitat was to be occupied) and one based on prey at K = 262–662 (Mladenoff et al., 1995, 1997). The Delisting Level was half of the habitat-based maximum (K = 500), and the Management Goal was a round number set at 53% of the prey-based maximum (K = 662); note in 2012 the wolf population was estimated above 815. That N_{goal} was set at or near K/2 suggests scientific reasoning linked to MSY approaches. The similarity of these bracketing values to MSY thinking is suggestive, although this was not stated explicitly until 2009 (see next section). Even if there is another explanation (besides MSY) behind the choice of N_{goal} and the Delisting Goal, such reasoning was not provided, implying that these parameters were deterministically generated by the model in **Figure 2**, rather than the product of value judgments.

Turning to how the science was conducted in the 1999 Plan and particularly in **Figure 2**, an unstated assumption and two assertions of fact were made by fitting the logistic growth curve to the annual wolf population estimates. The unstated assumption was that the changes in census methods in the winter of 1994–1995 and in the early 2000s (detailed in the **Supplementary Material**) were inconsequential, such that a single curve could be fit to the point estimates of population size in **Figure 2**. Those changes in census methods were designed and implemented by some of the authors of the 1999 Plan and 2007 Addendum to the Plan (**Supplementary Material**). Also, the curve represents a fact claim and a prediction, both of which required explanation that was not provided in the 1999 Plan. Specifically, the assertion was that density-dependent growth dynamics were occurring. The assumption of density-dependent population growth was not contrasted with alternative forms of growth nor tested explicitly.

Density-dependent dynamics were known to be common at the time, but far from universal in that a substantial proportion of wild animal populations did not show such dynamics (Fowler, 1987). The next publication did not clarify. A 2007 addendum to the 1999 Plan by the same authors stated "Van Deelen (unpublished) fit simple growth models to a XX [sic] year time series of wolf population estimates. Models fit were the discrete logistic model (CITATION) [sic]…" p. 7 (WDNR, 2007). The omissions indicated by [sic] in the quotation further clouded the issue.

Before the publication of the addendum (WDNR, 2007) to the 1999 Plan, Brook and Bradshaw (2006) explained why populations might not show density dependence, including: (a) substantial errors or changes in sampling or measurement can mimic or obscure density-dependent dynamics; (b) populations growing without spatial bounds and limited mainly by exogenous factors are not expected to show density-dependent dynamics; and (c) minimal changes in density over the sampling period might not produce such dynamics. All three scenarios might apply to Wisconsin's wolf population history, given that (a) census methods had changed twice and showed substantial variance among years, (b) the leading cause of death were by vehicles and poaching that were correlated to geographic spread and policies, respectively, not to density; moreover, density only increased over time slightly relative to measurement precision (Wydeven et al., 2001, 2004, 2009; Chapron and Treves, 2017; Treves et al., 2017c; Treves, 2019a; Santiago-Ávila et al., 2020). Therefore, the authors of the 1999 Plan had 8 years to make the foundation of **Figure 2** transparent and test its fact claims with mortality or reproductive data and sensitivity analyses. The prediction of future growth in **Figure 2** could have benefited from a scientific statement or estimate of uncertainty against the alternative of exponential growth, especially given that the two policy values discussed above lay in the prediction interval. Additional information came out in 2009, but the apparent value judgments underlying the wolf population model and the science used to buttress the state approach to target setting remained non-transparent, and still do today.

Transparency and Tests of Fact Claims From 2009 to 2020

In 2009, the authors of the state wolf population model published a book in which the authors clarified the fitting procedure for the logistic growth curve and the role of K in modeling. Furthermore, the model was used to explore a single scenario involving "maximum *sustained* yield" p. 150 (Van Deelen, 2009), emphasis added because no legal killing had yet taken place. This scenario was instead a prediction about future sustainability if regulated use was implemented. That 2009 chapter did not mention the changes in census methods.

The issue of change in census methods was obscured by errors in relating the history of Wisconsin wolf policy and monitoring in the first peer-reviewed article in 2015 (**Supplementary Materia**l). Finally, the 2009 book did not quantitatively evaluate the 1999 assumption of negative density dependence on birth or mortality, although some trends in density were presented (Wydeven et al., 2009). Other authors agreed that negative density dependence on wolf mortality was not apparent in the period 1995–2012 (Stenglein et al., 2015; Chapron and Treves, 2016, 2017). There was less agreement on other issues central to the state wolf population model and its assumptions.

(continued)

BOX 2 | (continued)

From 2016 to 2020, several teams of independent scientists began to raise questions about population dynamics and wolf mortality patterns. They found that evidence for negative density dependence in reproduction (Stenglein et al., 2015) was not transparently presented in quantitative format but instead depended on a crude line drawing (Chapron and Treves, 2017). Independent scientists reported that the census method was associated with the mortality hazard among radio-collared wolves, and the legal killing policies from 2003 to 2012 were associated with hazard and incidence of mortality among radio-collared wolves (Santiago-Ávila, 2019; Santiago-Ávila et al., 2020), both of which are exogenous factors that could reduce the influence of any density dependence. Furthermore, state reporting had appeared non-transparent about mortality patterns, the effect of census method on the means and variances of wolf population estimates, and the effects of actual wolf-hunting (2012–2014) on model predictions (Treves et al., 2014, 2017a,b,c; Treves, 2019a).

While the public policy process in our case study in 1999–2020 began transparently (i.e., the state announced it will hunt wolves), the conduct of the science was not fully transparent beginning with the 1999 Plan. Whether the authors of the 1999 Plan, its 2007 addendum, and 2009 book chapter began with MSY thinking or began instead with the assumption of density-dependent growth is unclear, but either way, the problem of using models as justification for policies that are likely derived from value judgments (in this case, valuing maximal yield) is illustrated.

The issue is broader than a scientific or political debate because US federal legal issues (and perhaps constitutional issues) are at play. Legitimate, competing value claims were not explicitly acknowledged, e.g., of Ojibwe sovereign tribes who view wolves as companions and equal to humans and federal treaty rights governing wolf hunting (David, 2009; Fergus and Hill, 2019), nor of diverse opponents of wolf hunting. We did not examine in detail those competing value claims here, but end by pointing out that that State of Wisconsin statutes proclaim that "legal title to, and the custody and protection of, all wild animals within this state is vested in the state for the purposes of regulating the enjoyment, use, disposition, and conservation of these wild animals." (WI STAT. ANN. §29.011).

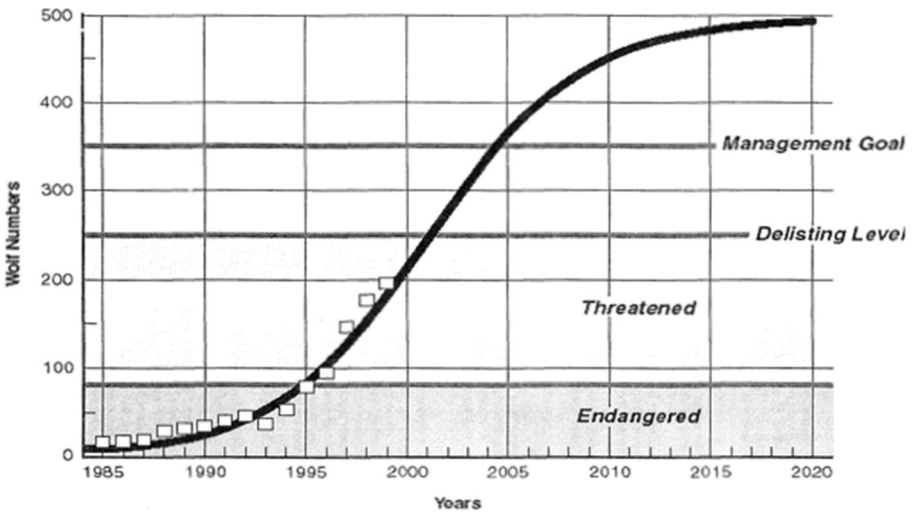


FIGURE 2 | The 1999 Plan Figure 7 "Wisconsin Wolf Population Growth If Carrying Capacity Is 500 Wolves." The 1999 Plan forecast wolf population growth to 2020 from a superimposed, generic logistic growth curve (WDNR, 1999). The model treated the population estimates as a single time series, although according to Treves (2019a) this should have been presented as two time series because of a change in wolf census methods in the winter of 1994–1995. The "Delisting Level" was set at 250, when the legal removal of wolves from the state's list of threatened and endangered species would begin. The "Management Goal" codified a population target ($N_{goal} = 350$), which is still the state population target today (USFWS, 2020). Vertical lines represented 5-year intervals and horizontal lines represented hundreds of wolves. Arguably, the explicit value judgments (Delisting Level and Management Goal) appear as outputs of the model, though their origin was not explained in WDNR (1999).

omission left a gap in understanding how a broader value-based approach could have occurred. The available evidence suggests that the gap was apparently filled implicitly (i.e., apparently through use of MSY reasoning) and introduced assumptions and untested fact claims about wolf census and density-dependent population dynamics) without fair scrutiny of alternatives, such as the population dynamic not being density dependent, or the carrying capacity not having been estimated accurately. A more reasoned approach would have been for the natural scientists to acknowledge the considerable uncertainty about the science and return to policy-makers (and ideally, social scientists and legal experts) for clarity on the value judgments being made.

NRM can attract controversy, often related to clashing values because in practice, NRM reflects long established belief systems, with varying scientific groundings and unknown or undisclosed risks. Wolves or perhaps endangered species, in general, may be particularly susceptible to implicit value judgments and contested science (Nie, 2001) but prior work has summarized how hundreds of management plans in which a claim was made of science-based harvest yet the basis in science was invisible to the public and to peer scientists seeking information from managers (Artelle K. A. et al., 2018). Some of these cases might fit the descriptions of "political populations" (Darimont et al., 2018), i.e., characterized by unrealistic parameters to promote

a particular policy. Scientists working outside of government agencies are also implicated in non-transparency about so-called sustainable-use models. Scott et al. (2007) found policy advocacy throughout most of a random selection of wildlife papers from noted scientific journals, to varied extents, and with different degrees of disclosure. A recent analysis of publications regarding trophy hunting revealed non-governmental researchers evincing breaches of transparency (Koot et al., 2020), including omission of methods, failure to disclose competing interests, reinforcing weak evidence by publishing it repeatedly, and ignoring contradictory evidence but citing allied authors despite the weakness of their evidence (see also Santiago-Avila et al., 2018a; Treves et al., 2019). Therefore, we hypothesize that wildlife science is experiencing its own reproducibility crisis like that of many other scientific disciplines (Open Science Collaboration, 2015; Allison et al., 2016; Baker and Brandon, 2016; Goodman et al., 2016; Munafò et al., 2017; Clark and Alvino, 2018), but with potential consequences for the persistence of wild populations.

Scientists allied to agencies might face a competing interest to conceal potential problems with sustainability from public view and the scrutiny of independent scientists. The competing interest might be if agencies allocating natural resources to current users are under political pressure to allocate more than is safe. The wolf case study is one example where the 1999 estimate of carrying capacity of wolves was likely 25–50% lower than current estimates, the inference being that the MSY-associated population goal was set too low for the safety of the wolf population from unsustainable use. Non-transparent science, fact claims that are never tested, and unwarranted assumptions all characterized that case. Should these processes be widespread, we see a possible explanation for why natural resource management has a record of poor performance globally.

RECOMMENDATIONS FOR NATURAL RESOURCE MANAGEMENT MODELS AND SUSTAINABLE USE POLICIES

To guard against the misappropriation of scientific methods and inferences, and to promote a clearer understanding of the rationale for the management decisions made by public officials, we argue for enhancing the level and scope of scientific integrity practiced in NRM science and fortifying it with rigorous independent review of extraction policies (**Figure 3**).

Recommendation 1 is aimed at the dialogue between science and policy. We encourage a clear demarcation between scientists' value judgments and scientific observations or inferences when providing scientific communications. For example, one might state "We ought to do x or y because we believe those actions are right…" (values) and separately, "we predict the outcomes will benefit actor a and harm actor b" (fact-claims). Those two paired statements make obvious the need for science in service to public policy without cloaking value judgments in fact claims. In turn, the scientists serving the public should state assertions along the lines of, "The net costs and benefits of x and y to actors a and b are…, and the side-effects, long-term consequences, and unnamed actions and actors might be…." The latter statement does not conceal the value judgments themselves, which science cannot evaluate with observations or inferences about how the system works, nor does it ignore unspecified actions, actors, or effects. If those were ignored, the scientists would have unjustly limited their inquiry because science can and should make the public more aware of unanticipated effects and "winners and losers" without interposing the scientists' preferences unduly. Thus, science can inform values and lead to management shifts if the previously unappreciated actors, effects, or actions are perceived as better or worse than the preconceived ones in policy statements.

Recommendation 2 is also aimed at the dialogue between scientists and NRM decision makers. Scientists should be alert to hidden value judgments at each step (Doremus, 2004). For one, we advise scientists not to begin with "maximum sustainable yield" as their own *de facto* starting point or end goal because it camouflages a value judgment within scientific estimates and predictions yet to be validated. A corollary is to present transparently more than one scenario to the public and to policy-makers. The additional scenarios should reflect sensitivity analyses (What if K is much higher or lower? What if natality or mortality are linearly related to density or unrelated?) and risk assessments (How certain are we about our pronouncements? What if conditions change?). Indeed, we recommend higher standards of evidence in modeling sustainability before authorizing extraction. However, we suspect the politics behind the use of wild populations will not abide such delays. Instead, we advise that extraction models be adapted rapidly when changes to allowable uses are proposed. We acknowledge that agreement might be needed from the scientific community on standards of evidence for natural resource extraction under various circumstances (Oro, 2013), as in other fields (van Eeden et al., 2018; Treves et al., 2019). That agreement should address how scientists should respond with precaution when risk or uncertainty is too great to recommend even moderate extraction (Doremus, 2004). Likewise, we suggest that the dependence of MSY on biological estimates (e.g., of carrying capacity, growth rates, and density dependence) be made explicit in scientific communications, so that changes in biological parameters or parameters of human use result in changes in MSY estimates, rather than fixing population goals and quotas for longer periods. Finally, we encourage policy makers to engage with social scientists and legal scholars, who can appropriately characterize the diversity of values among the public and policy makers (Manfredo et al., 2019; Darimont et al., 2021), so that influence of values can be explicitly acknowledged (Doremus, 2004).

This recommendation may require retooling of governmental procedures to support its strictures. Many regulatory processes, under which decisions regarding natural resource extractions are made, are not well suited to the analysis of mutually exclusive goals or alternative scenarios. We point to laws, such as the US National Environmental Policy Act (which entails analyses regarding alternative actions, including a "no action" option) as potential models. While a preferred alternative may still be advanced, analysis of a range of options more transparently reveals the consequences of our choices on both the natural

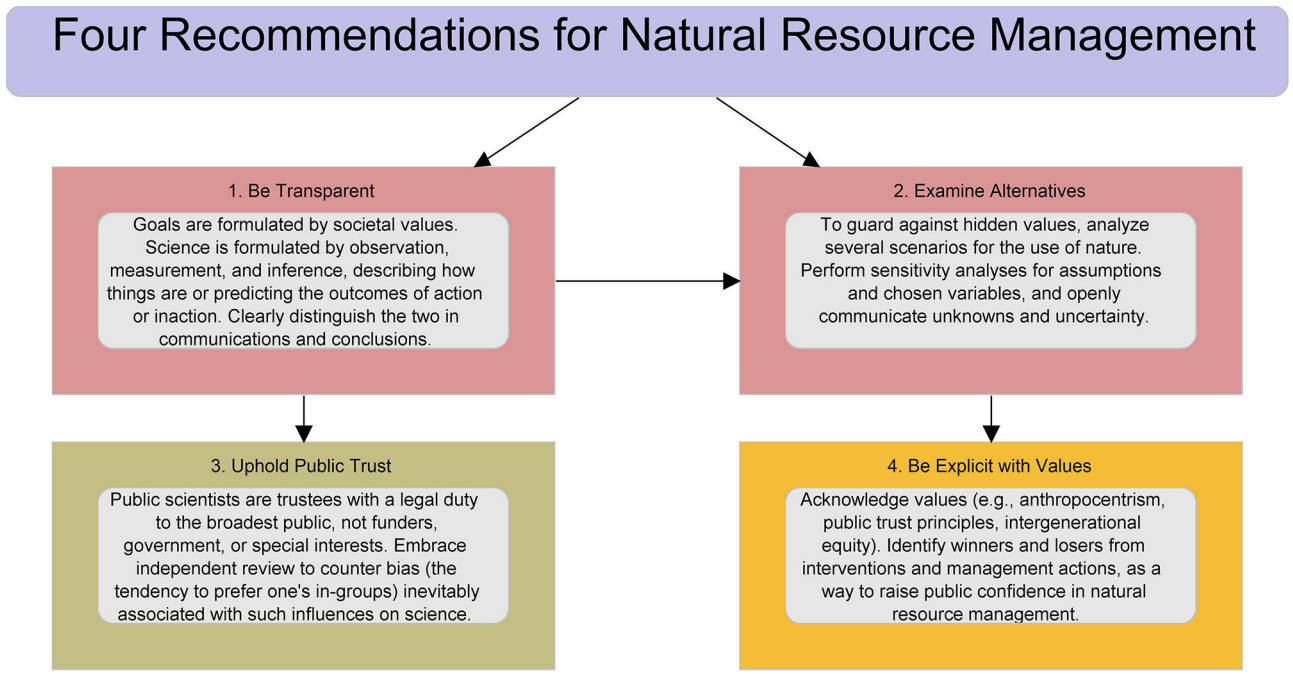


FIGURE 3 | Recommendations for where the science begins and ends in NRM. Pink boxes are aimed at the dialogue between science and public policy, the orange box is aimed at policy-makers, and the green box is aimed at scientists. Arrows indicate cause and effect relationships, e.g., being transparent and examining alternatives will help scientists to uphold the public trust although not sufficient by themselves.

resources themselves and the involved interests (both human and non-human). Alternatives can be better framed in terms of the value judgments that underlie them, and thereby help the public and decision makers to discern among competing policy goals.

Avoiding risk, especially for vulnerable parties, is often an ethical standard for well-reasoned action, regardless of the uncertainty or magnitude of the risk measured by scientists. Risk and uncertainty can be characterized or quantified and should be disclosed, which can be the foundation of a more informed decision and can help efforts to be adaptive in the future (Doremus, 2004; Regan et al., 2005; Artelle et al., 2013; Chapron and Treves, 2016; Milner-Gulland and Shea, 2017). Such clarity might also motivate efforts to increase the quality of ecosystems so that carrying capacity would be increased for populations facing human use, which might favor preservation for the future. The latter is a value judgment consistent with public trust thinking described below and also consistent with some indigenous systems of values and associated management practice.

Our third recommendation is that the vital role of independent review deserves amplification (Allison et al., 2016). Independent reviewers can help scientists see the alternatives and make them more than straw-person hypotheses. Also, diversity within the groups of scientists working on a problem may enhance the transparency of assumptions held by a smaller number of scientists who have been trained similarly, influenced each other over long periods, or face similar incentives for particular methods or ways of interpreting observations. Indeed, the values held by scientists within US wildlife management agencies often do not reflect that of the broader public (Manfredo et al., 2019). The legal doctrine of the public trust could provide an important guide to NRM policy-makers and managers when their personal values do not match the majority of the broad public (Treves et al., 2017b). It forms the basis for guidance to the judicial, legislative, and executive branches of democratic governments that declares that components of the environment are held in trust for the broadest public, including current and future generations. That legal doctrine has been interpreted by many to imply a thorough and stringent transparency by the trustees (e.g., agencies and individual staff) as a legal duty to account explicitly and in a sophisticated manner for uses and preservation of the trust components, particularly when allocating public assets to private users, as is usually the case with NRM (Sax, 1970, 1971, 1980–1981; Wood, 2009, 2013; Blumm and Wood, 2017; Treves et al., 2017b, 2018a). This legal tradition stands in stark contrast to interpretations that professional societies and agencies should take sides with extractive user groups, often in opposition to private property and animal rights interests, e.g., synopsis in Batcheller et al. (2010). Trustees are called to avoid practices that promote picking their preferred scientists to cite or cherry picking their preferred scientific findings and to reject the notion that governments can vaguely aspire to natural resource sustainability without enforcing measurable goals or enforcing against illegal uses (Nie et al., 2020). Precautionary decision making, intergenerational equity, and public trust thinking of the sort we recommend above are not new. Sharing a concern for future generations' needs and non-anthropocentric concerns

for other beings, contemporary indigenous governance systems are notable in our context of transparently disclosing the role of values in their governance (Artelle K. et al., 2018). For example, many such NRM governance documents begin by stating underlying and guiding principles (e.g., Te Runanga o Kaikoura, 2007; Marine Planning Partnership Initiative, 2015; Fergus and Hill, 2019), often with a cultural context that brings meaning to the analysis as a whole.

Interest group relationships to scientists can potentially lead to corruption if interest groups can capture agencies, and agencies, in turn, control funding that can capture scientists. Alliances or rivalries between agencies and scientists are, by definition, competing interests that commonly bias science and often lead to irreproducible results (Munafò et al., 2017). NRM is not revenue-neutral so the policy neutrality of scientists who may stand to benefit from extraction is paramount—particularly regarding the analysis of winners and losers. In North America, wildlife agencies are commonly and unduly responsive to a minority of extractive interest groups (Gill, 1996; Batcheller et al., 2010; Clark and Milloy, 2014; Treves et al., 2017b; Serfass et al., 2018), so the scientists who support agency plans frequently encounter competing interests and become suspect by the public and peers of biased results (Doremus, 2004, Rohlf, 2019). Stringent attention to principles of scientific integrity (National Academies of Science, Engineering and Medicine, 2017) and following long-standing and novel safeguards for reproducibility (Allison et al., 2016; Munafò et al., 2017) would go a long way to avoiding a full-blown reproducibility crisis and losing public confidence in the field of sustainable use altogether. Accordingly, we recommend that natural resource managers seek authentic independence in their reviewers, which implies data sharing, replication of findings, tests of cherished assumptions, and avoiding affiliative or antagonistic relationships with any scientists. Separating funding for science from the policy-making arms of the government would also help if independence between the two arms were enforced.

The above relates to unstated and broader value judgments about human claims to nature. Therefore, our final recommendation relates to transparent claims and assessments of winners and losers. The terms and techniques of sustainable use and MSY expose an underlying anthropocentric paradigm. Treating living organisms as objects that can be "owned," subject to "harvest," or the amount such extraction "yields" enfolds value judgments about the priority of humans over non-humans. These terms are not scientific terms but rather euphemisms (Johns and DellaSala, 2017) and privilege the worldviews of some people over those of others, e.g., settler societies over Indigenous peoples (Eichler and Baumeister, 2018). Indeed, much of sustainable wildlife management is simply wildlife demography and ecology, but in the service of anthropocentric interests. We are not asking that management documents expound detailed philosophical discourses on values. Rather, they should simply and coherently state their basic assumptions and the views on which their decisions are based. For example, "we hold non-anthropocentric worldviews that all of nature is held in trust for the futurity of all life on Earth" (an approximation of many of the present authors' worldviews). If the value judgments are stated clearly and allow the general, non-scientific public to discern what is evidence and what is value based, the discussion has improved in our view and that of others who have taken deep looks at NRM and endangered species policy (Doremus, 2004; Carroll et al., 2017; Nie et al., 2020).

In conclusion, we believe that scientific integrity is a fundamental professional ethic and, in many cases, a legal obligation of scientists. Public scientists or government trustees who compromise scientific integrity open themselves to capture by narrow interest groups vying over permit fees or private uses of nature (Finley, 2011; Kolowich, 2016; Treves et al., 2017b). Our cautions are particularly focused on public scientists—individual scientists supported by public monies, whether by past training, salaries, or current project grants. We encourage strict adherence to transparency which demands introspection by scientists first and then clear, honest communication to all others.

AUTHOR CONTRIBUTIONS

AT wrote the first draft. All authors contributed equally to the conception of the study and contributed revisions.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.631998/full#supplementary-material

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Decision-Making for Rewilding: An Adaptive Governance Framework for Social-Ecological Complexity

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Rewilding can be defined as the reorganisation or regeneration of wildness in an ecologically degraded landscape with minimal ongoing intervention. While proposals for rewilding are increasingly common, they are frequently controversial and divisive amongst stakeholders. If implemented, rewilding initiatives may alter the social-ecological systems within which they are situated and thus generate sudden and unforeseen outcomes. So far, however, much of the discourse on the planning and implementation of rewilding has focused on identifying and mitigating ecological risks. There has been little consideration of how rewilding could alter the human components of the social-ecological systems concerned, nor governance arrangements that can manage these dynamics. This paper addresses this gap by proposing a generic adaptive governance framework tailored to the characteristics of rewilding, based on principles of managing complex social-ecological systems. We integrate two complementary natural resource governance approaches that lend themselves to the contentious and unpredictable characteristics of rewilding. First, adaptive co-management builds stakeholder adaptive capacity through iterative knowledge generation, collaboration and power-sharing, and cross-scale learning networks. Second, social licence to operate establishes trust and transparency between project proponents and communities through new public-private partnerships. The proposed framework includes structural and process elements which incorporate a boundary organisation, a decision-into-practise social learning exercise for planning and design, and participatory evaluation. The latter assesses rewilding outcomes and pre-conditions for the continuation of adaptive governance and conservation conflict resolution.

Keywords: adaptive capacity, adaptive co-management, conflict transformation, conservation conflict, livelihoods, knowledge, social licence to operate, partnership

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INTRODUCTION

Rewilding, defined in this paper as the reorganisation or regeneration of wildness in an ecologically degraded landscape with minimal ongoing intervention, is a novel and rapidly developing conservation concept, with a burgeoning number of initiatives proposed or implemented (Pettorelli et al., 2019) in diverse social and ecological contexts (Butler et al., 2019). Rewilding initiatives can

be contentious and divisive amongst the multiple stakeholders involved, creating conflicts that can limit its effectiveness. After implementation, rewilding can also generate sudden and unforeseen ecological changes (Corlett, 2016), and hence unexpected benefits and costs for the stakeholders involved (Pettorelli et al., 2018).

Social-ecological systems consist of societal and ecological components in mutual interaction. They are typified by four key characteristics: interlinked scales and components; non-linear dynamics caused by cross-scale reinforcing feedback loops that amplify interactions; emergence of sudden and unexpected outcomes; and thus irreducible uncertainty (Gallopin, 1991). Plummer and Armitage (2007) suggest that decision-makers must focus on two primary outcomes from the stewardship of social-ecological systems: ecosystem condition and sustainable livelihoods. In terms of rewilding, decision-makers must anticipate that any initiative is, as rewilding implies, likely to alter existing relationships between system components, potentially generating unanticipated ecosystem and livelihood outcomes.

However, there has been little analysis of rewilding from a social-ecological perspective, or consideration of how to manage the stakeholder conflicts and uncertainties that could emerge (Butler et al., 2019; Durant et al., 2019; Drouilly and O'Riain, 2021). Instead, much of the discourse on planning and implementing rewilding has focused on identifying and mitigating ecological risks (e.g., Batson et al., 2015; Robert et al., 2015; Nogués-Bravo et al., 2016). As a result, many rewilding initiatives are undermined by social rather than ecological challenges (Coz and Young, 2020; Drouilly and O'Riain, 2021).

Governance of natural resources can be defined as "the norms, institutions and processes that determine how power and responsibilities over natural resources are exercised, how decisions are taken, and how citizens… participate in and benefit from [their] management" (Campese et al., 2016, p. 1). Adaptive governance is necessary for social-ecological systems due to their dynamic and unpredictable characteristics. In general, it involves flexible, polycentric and self-organising institutions that link across a system's scales, thus allowing suites of co-ordinated responses to complex challenges at the necessary levels. Two key attributes of adaptive governance are learning networks that promote knowledge generation and exchange amongst stakeholders across scales, and "bridging organisations" or individuals that broker and facilitate these networks (Folke et al., 2005).

Given the experimental nature of rewilding, and its potentially contentious and unpredictable influences on social-ecological system dynamics, we argue that adaptive governance should be central to both its planning and implementation. In this paper we consider rewilding from a social-ecological systems perspective, and in particular the governance models required to steward the inevitable shifts in human-nature relationships. We propose the integration of adaptive co-management (ACM) and social licence to operate (SLO) in a generic governance framework for rewilding initiatives. We explore why and how this approach could form a foundation for more effective planning and management of rewilding initiatives. We believe that the proposed framework could support the implementation of the newly adopted resolution of the International Union for the Conservation of Nature (IUCN) on rewilding, which aims to develop guidelines for rewilding that include assessments of the relative risks and rewards to ecosystems and local communities (IUCN, 2021).

ADAPTIVE CO-MANAGEMENT

The design of governance approaches for complex social-ecological systems is a growing field of research (e.g., Folke et al., 2005; Armitage et al., 2009; Plummer et al., 2017). ACM has recently evolved as an effective refinement of adaptive governance. It combines the iterative co-learning, knowledge generation and problem-solving of adaptive management with the stakeholder collaboration, power-sharing and alternative institutions of co-management (Armitage et al., 2009; Berkes, 2009; Keith et al., 2011). Folke et al. (2002, p. 8) define ACM as "a process by which institutional arrangements and ecological knowledge are tested and revised in a dynamic, ongoing, self-organised process of trial-and-error." ACM is advocated for the stewardship of social-ecological systems because it encourages cross-scale social networks, integration of multiple knowledge types to solve complex and unprecedented problems, and reflexivity through continual evaluation and learning, which together enhance decision-makers' capacity to anticipate uncertainty and respond to shocks (Olsson et al., 2004; Armitage et al., 2009; Fabricius and Cundill, 2014).

While the "what" of ACM is clear, it has been critiqued for the lack of detail on the "how" and limited evidence of clear outcomes (Rist et al., 2013; Fabricius and Cundill, 2014; Plummer et al., 2017). This is understandable because ACM is itself an emergent property of a social-ecological system, often occurring in response to an exogenous shock or resource crisis (e.g., Olsson et al., 2004, 2006; Butler et al., 2008; Plummer, 2009; Cox et al., 2020). Consequently, there is no blueprint for the process and/or outcomes of ACM since each instance will be context-specific and self-organising (Plummer et al., 2012). Nonetheless, ACM can be engineered by creating a structure and process founded on its principles of multi-stakeholder engagement and learning (e.g., Cundill and Fabricius, 2010; Smedstad and Gosnell, 2013; Butler et al., 2016a,b).

Despite the reasonably recent implementation of ACM, there are already examples of this approach successfully mitigating conservation conflict amongst stakeholders, for example regarding dugong hunting (Butler et al., 2012), seal tourism and salmon fisheries (Butler et al., 2015a; Bellanger et al., 2020; Cox et al., 2020) and "hard edges" around protected areas (Plummer et al., 2017). These examples have identified key pre-conditions for the maintenance of conflict resolution, including long-term government support for the process, strong leadership and champions, bridging organisations or individuals, and cross-scale partnerships (Young et al., 2012; Butler et al., 2015a; Cox et al., 2020). These are now being mainstreamed into conservation conflict efforts (e.g., Redpath et al., 2013; Young et al., 2016; Redpath et al., 2017), where the focus is shifting from conflict resolution, which emphasises compromise and jointly

agreed outcomes, to conflict transformation, which leverages stakeholder concern and engagement in contentious issues to transform systems (Skrimizea et al., 2020).

SOCIAL LICENCE TO OPERATE

SLO emerged in the 1990s to describe the informal acceptance, approval or trust that a local community extends to a corporate entity or industry developing new operations, with a specific application to mining (Lacey and Lamont, 2014). The concept has since been extended to other industries, such as forestry (Moffat et al., 2016). SLO is useful for governance because it highlights the need for development proponents to acknowledge and address social concerns about a novel proposal and is the starting point for dialogue between stakeholders (Moffat et al., 2016). It also emphasises the need for a relationship based on trust and transparency to be cultivated between the proponents and local communities, and hence ethical governance and social justice (Lacey and Lamont, 2014). SLO implies that an agreement will be reached between a developer and communities which mirrors the "license" granted by government to the developer to undertake operations, with its necessary safeguards (Moffat et al., 2016).

Kendal and Ford (2017) have assessed the relevance of SLO to threatened species programs. Conservation interventions are likely to be more complicated than a development intervention because stakeholders tend to range from local to global and have a greater spectrum of attitudes on environmental issues (Ford and Williams, 2016). Because conservation initiatives are usually government-led and therefore acting in the public rather than the private interest, more complex partnerships are required between the public sector and local stakeholders (Ojha et al., 2016). Regardless, SLO is appropriate for conservation purposes because it emphasises the need for practitioners to develop trusting relationships with local and other participants, to recognise and address the diversity of their views, and to anticipate and address potential conflict through transparent governance processes (Kendal and Ford, 2017).

AN INTEGRATED ADAPTIVE GOVERNANCE FRAMEWORK

ACM and SLO provide over-lapping and complementary themes that could contribute to the improved adaptive governance of rewilding. ACM provides a specific focus on iterative co-learning, knowledge generation, and cross-scale networks. It also highlights the need for leadership and the roles of bridging organisations or individuals to facilitate these processes. SLO emphasises the establishment of trust and transparency between project proponents and communities, and the formation of novel public-private partnerships amongst multiple stakeholders. Common to both approaches are stakeholder partnerships across scales, recognition of the diversity of their views, social justice, equal representation and power-sharing, new institutional arrangements, and conflict resolution aided by these principles. Our proposed framework for governing rewilding integrates these themes through two elements: structure and process.

Structure

The core structural element is a facilitation team which acts as a boundary organisation amongst the multiple private and public stakeholders across scales of the system. It should be emphasised that the establishment and maintenance of a facilitation team requires adequate and consistent resourcing, something which is often overlooked by funders (Butler et al., 2016a). The facilitation team identifies and engages stakeholders, organises activities that enable dialogue and consensus-building, brokers knowledge and information, and mediates in conflict. The team must be regarded as independent, credible and trustworthy by all stakeholders (Olsson et al., 2004; Armitage et al., 2009; Cundill and Fabricius, 2010), and act as a conduit between them, creating the learning networks that are critical to harnessing knowledge and generating innovation (Olsson et al., 2004). Hence, team members must be skilled in cross-sectoral communication, mediation, conflict resolution, event organisation and facilitation (Butler et al., 2017). In light of the importance of the facilitation team, the appointment process is crucial, as is the need for a grievance process to allow communities to voice any concerns.

The team's first task is to carry out a stakeholder analysis for the rewilding location and its social-ecological system. There are numerous suitable methodologies (e.g., Schultz et al., 2007; Reed et al., 2009; Baird et al., 2014), but particular attention should be paid to power relationships amongst stakeholders and communities, which are often overlooked (Armitage et al., 2009; Fabinyi et al., 2014; Butler et al., 2015b; Boonstra, 2016). To understand potential power asymmetries in any rewilding project, it would be essential to analyse the political dynamics, and to ensure that weaker or marginalised stakeholders are adequately represented. Additionally, the most powerful must be willing to share decision-making, rather than dominate it. It may also be necessary to create a steering committee, independent from the facilitation team, which represents the major stakeholder groups to provide the political legitimacy for the governance structure, and to formally link to national policy processes (Butler et al., 2016a,b).

Process

Our over-arching process is the well-known adaptive management cycle, involving the steps of plan, design, implement, monitor and evaluate, and revise (Williams et al., 2009). We simplify this into three steps (see **Figure 1**):

Step 1: Plan and design. This applies to the plan and design of the rewilding initiative. There may be legal requirements which pre-determine the format of this activity, particularly where public lands such as national parks are concerned, or locations including First Nation or Aboriginal land rights (Pratt Miles, 2013). Encouraging stakeholders to participate, and understanding their incentives to do so, can be problematic, and contains its own ethical and political tensions (Cooke and Kothari, 2001; Stringer et al., 2006; Hurlbert and Gupta, 2015) which the facilitation team must have skills to manage (Butler et al., 2017). However, fundamental is the creation of a forum that

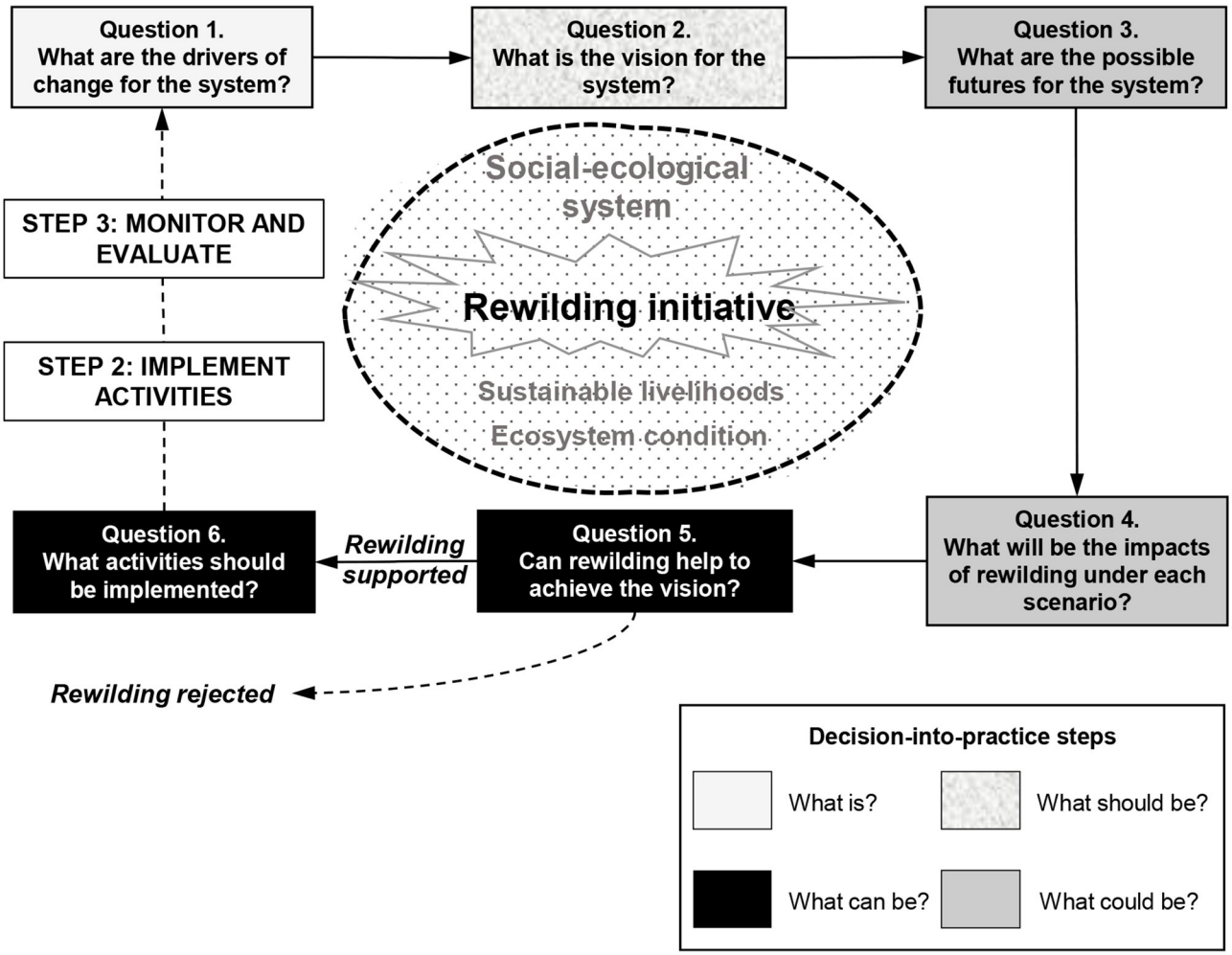


FIGURE 1 | The sequence of questions (1–6) involved in the decision-into-practise exercise proposed for Step 1 (plan and design). If the rewilding initiative is supported after Question 5 then the outcome of Question 6 leads to Step 2 (implement activities) and Step 3 (monitor and evaluate). The process then cycles back into Step 1 and the sequence of questions is repeated in subsequent adaptive management cycles.

can engage stakeholders in open dialogue, and where different knowledge can be considered and respected equally.

Step 1 could be initiated with a multi-stakeholder activity which catalyses social learning and consensus-building. Brown (2008) designed a "decision-into-practice" learning exercise which has been effectively adapted to initiate similar planning processes for community development (Brown and Lambert, 2015) and climate change adaptation (Butler et al., 2015b, 2016c). Referring to the system and issue concerned, four questions are addressed in succession: "what is?" "what should be?" "what could be?" and "what can be?" resulting in an agreed set of actions. In this case the four questions are expanded to six, and the issue is the potential effects of a rewilding initiative within a social-ecological system, and consideration of potential ecosystem and livelihood outcomes (**Figure 1**).

Question 1 addresses the drivers of change influencing the system, thus establishing the social-ecological context and "what is?" (**Figure 1**). This deliberately identifies multiple social (e.g., human population trends, livelihood changes) and ecological (e.g., climate change, habitat dynamics) drivers that rewilding will interact with. Question 2 establishes the stakeholders' vision for the system, and hence a consensus on "what should be?" including the role and impact of conservation on material, social and subjective aspects of human wellbeing (De Lange et al., 2016; Woodhouse et al., 2016). Question 3 examines potential future system states given trends and uncertainties in the primary drivers identified in the first question. Scenario planning is an effective and well-established tool for this activity (e.g., Oteros-Rozas et al., 2015; Butler et al., 2020).

The process then casts the rewilding initiative into the system context and stakeholders' agreed vision (**Figure 1**). Question 4 considers the potential impact of rewilding on each future system state. At this stage various tools and information already established in rewilding and restoration science could be applied, including landscape suitability assessments, prey availability (for carnivores), and current management effectiveness. Based on

these assessments, Question 5 judges whether the initiative complements or impedes the attainment of the stakeholders' vision, thus asking "what can be?"

If rewilding is compatible with the vision, or requires modification, Question 6 seeks to agree a program of strategies and innovations which can be rolled out in Step 2 (below). At this stage, agreement can be reached about identifying potential social risks to monitor, together with their baselines. Whilst such risks will vary according to location and populations, Woodhouse et al. (2016) have developed generic indicators of social risks or outcomes of conservation, which could be used to help identify social components to measure. If rewilding is not compatible with the vision and is therefore not supported, the proposal could be rejected at this point. Importantly, this co-learning process may still galvanise stakeholder action to better govern the existing system toward an agreed vision.

TABLE 1 | Proposed indicators for evaluating (A) rewilding governance outcomes and (B) pre-conditions for ongoing adaptive governance, showing alignment with ACM and SLO themes, adapted from Butler et al. (2015a).

Indicator	Governance themes
A. Outcomes	
1. New institutional arrangements	New institutions–ACM, SLO
2. New institutions formally codified	New institutions–ACM, SLO
3. *Rewilding management plan*	New institutions–ACM, SLO
4. Legitimisation of policies and actions	New institutions–ACM, SLO
5. Changes in perceptions and actions	Iterative co-learning–ACM
6. Engagement and learning across scales	Cross-scale networks–ACM
7. Questioning of routines, values and governance	Iterative co-learning–ACM
8. Creative ideas for problem-solving	Knowledge generation–ACM
9. Agreed upon sanctions	New institutions–ACM, SLO
10. No party asserting its interests to the detriment of others	Power-sharing–ACM, SLO
11. *Rewilding outcomes (including social outcomes) acceptable to all parties*	Power-sharing, sustainable livelihoods–ACM, SLO
12. Acceptable level of ecosystem function	Power-sharing, ecosystem condition–ACM, SLO
B. Pre-conditions	
1. Presence of a bridging organisation or individual	New institutions–ACM
2. Commitment to long-term institution building	New institutions–ACM, SLO
3. Adaptable portfolio of management resources	Knowledge generation–ACM
4. Provision of training and capacity-building	Knowledge generation–ACM
5. Stakeholders drawing on and sharing diverse knowledge	Knowledge generation–ACM
6. Formal and regular evaluation with stakeholders	Iterative co-learning–ACM
7. High quality of information and resources	Knowledge generation–ACM
8. Leaders prepared to champion the process	Leadership–ACM
9. Supportive policy environment	Power-sharing–ACM, SLO
10. Transparency of stakeholders' goals and values	Trust and transparency–SLO
11. Trust amongst stakeholders	Trust and transparency–SLO
12. Participation of all impacted stakeholders	Representation–ACM, SLO

Indicators added specifically for rewilding are italicised.

Step 2: Implement activities. Here we refer to the activities identified by Step 1. Fundamental to this is multi-stakeholder engagement in learning-by-doing experiments (Armitage et al., 2009; Plummer, 2009; Plummer et al., 2012). Each may involve a sub-set of actors, and possibly others additional to those identified in Step 1.

Step 3: Monitor and evaluate. This should be engrained within all activities to create a culture of ongoing reflection and learning (Armitage et al., 2009), enabled by the facilitation team and championed by leaders. Different forms of monitoring and evaluation may be applied to different aspects of the initiative. For example, an overall Theory of Change (ToC) could be developed for the rewilding initiative which articulates a vision of change, and systematically describes the sequence of activities, outputs, outcomes and impacts to achieve it, and the assumptions about the relationships between interventions and change (Vogel, 2012; Bours et al., 2013; Maru et al., 2018). If the ToC is carried out in a participatory process which engages stakeholders to reflect and learn, this has the added advantage of catalysing action to improve the ongoing design in subsequent cycles (Butler et al., 2015a, 2016a; Plummer et al., 2017; Trimble and Plummer, 2018; Cox et al., 2020).

Any evaluation should also consider the effectiveness of the governance process and necessary adjustments in terms of ACM and SLO principles. Plummer and Armitage (2007) devised a framework to measure ACM outcomes in terms of sustainable livelihoods and ecosystem condition, plus processes and institutions, that could be expanded to incorporate non-material aspects of human wellbeing. Armitage et al. (2009) identified further pre-conditions for the continuation of effective ACM. These frameworks, and methods for applying them have since been trialled in different natural resource management (e.g., Cundill and Fabricius, 2010), protected area (e.g., Plummer et al., 2017), climate adaptation (e.g., Butler et al., 2016b) and conservation conflict contexts (e.g., Butler et al., 2015a; Cox et al., 2020). The primary outcome sought through SLO is community agreement and acceptance of an initiative, exhibited as degrees of developer-community partnerships, trust, transparency and conflict resolution. While it is recognised that monitoring these outcomes is important (Roche and Bice, 2013), and SLO indicators have been developed for management (e.g., Boutilier et al., 2012; Provasnek et al., 2017; Lindman et al., 2020), their focus has been cost-benefit assessments, and as yet no governance-focussed frameworks exist for evaluating SLO.

Considering the complementarities between ACM and SLO's themes, an indicator framework is suggested which assesses institutional, process, wellbeing, livelihoods and ecosystem

outcomes, and pre-conditions for adaptive governance to continue (**Table 1**). This adapts an approach originally designed for evaluating the ACM of conservation conflict by Butler et al. (2015a). To apply the framework for rewilding, we have refined two outcome indicators: "rewilding management plan" and "rewilding outcomes (including social outcomes) acceptable to all parties" (**Table 1**). Tools such as community surveys, which are often applied for SLO (Roche and Bice, 2013) could be applied to assess the latter. To evaluate conflict transformation, which capitalises on the identification of the root socio-political sources of conflict, the outcome indicator "questioning of routines, values and governance" could examine stakeholders' underlying perceptions of the drivers of conflict, and whether the process has succeeded in altering them.

DISCUSSION

Rewilding is emerging as a pragmatic approach to repairing damaged ecosystems, yet guidelines for its governance remain relatively immature. Despite the critical importance of stakeholder engagement, adaptive experimentation and learning, the landmark IUCN/SSC guidelines (IUCN/SSC, 2013) only advise proponents to consider community perceptions, costs, and benefits. They do not incorporate adaptive governance approaches that are tailored to the social-ecological reverberations that rewilding is likely to induce. We have attempted to address this gap by proposing a governance approach based on contemporary principles of ACM and SLO. We hope to have broadened the focus of rewilding from a discussion tightly focused on conservation biology and conservation objectives to include social-ecological systems thinking, including human wellbeing, sustainable livelihoods, and ecosystem condition outcomes. We would argue that whilst this may add extra layers onto a rewilding initiative, the risks of not incorporating these social dimensions at the outset could result in rewilding initiatives becoming hampered by long-term and acute conflicts, with negative impacts on biodiversity and human wellbeing.

We believe that our approach is sufficiently generic to be applicable across the diversity of contexts in which rewilding is being considered. Since both ACM and SLO are themselves evolving, and have not yet been applied to rewilding, there is no blueprint for their application. Armitage et al. (2009) suggested that where resource use is poorly defined or distributed over large geographical areas with a plethora of stakeholders, and hence high transactional costs, ACM may be less effective. This has been experienced in the UK (Butler et al., 2008; Young et al., 2010) and may limit the utility of our framework in some contexts. Consequently, our proposed structure and process is not prescriptive, and deliberately only offers a skeleton to be tested. Not all elements need to be addressed in-depth in all rewilding initiatives—indeed, our intention with this framework is not to drain stakeholder energy from the rewilding activity, but rather to pre-empt and manage potential conflicts that might hamper the effectiveness of initiatives. As such, the indicator framework proposed in **Table 1** for monitoring and evaluating rewilding outcomes and pre-conditions for ongoing adaptive governance contains the key themes of our approach, which should be maintained if possible. Trials of the approach would iteratively inform future initiatives and streamline its structure and process.

A review of rewilding case studies indicates that some rewilding initiatives are evolving adaptive governance (Butler et al., 2019). We suggest, however, that our approach should be engineered in advance, as has been achieved for ACM in some cases (Cundill and Fabricius, 2010; Smedstad and Gosnell, 2013; Butler et al., 2016a,b). This could avoid significant transaction costs in controversial initiatives, where stakeholder conflict may otherwise escalate. In less contentious cases, our approach would still promote transparent governance and adaptive capacity and enable stakeholders to attain livelihood and ecosystem outcomes while accounting for future uncertainties. Even if during Step 1 (plan and design) a rewilding proposal is not supported, the process may still catalyse improved stewardship of the social-ecological system concerned. Whatever the approach, to be effective any adaptive governance process requires adequate and sustained resourcing, including support for bridging organisations or individuals.

In conclusion, we suggest that the proposed adaptive governance framework can accommodate the emergent uncertainties and conflicts characteristic of a social-ecological system that is altered by rewilding. We thus call on decision-makers and practitioners to test our suggested structure and process, including the application of our evaluation indicators.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

JRAB, MM, NP, SMD, JTT, and JCY all contributed to the development of the concepts presented in the paper. JRAB led on the writing, supported by MM, NP, SMD, JTT, and JCY. All authors contributed to the article and approved the submitted version.

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Living With Elephants: Evidence-Based Planning to Conserve Wild Elephants in a Megadiverse South East Asian Country

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Theory of Change (ToC) and Social Return of Investment (SROI) are planning tools that help projects craft strategic approaches in order to create the most impact. In 2018, the Management & Ecology of Malaysian Elephants (MEME) carried out planning exercises using these tools to develop an Asian elephant conservation project with agriculture communities. First, a problem tree was constructed together with stakeholders, with issues arranged along a cause-and-effect continuum. There were 17 main issues identified, ranging from habitat connectivity and fragmentation, to the lack of tolerance toward wild elephants. All issues ultimately stemmed from a human mindset that favors human-centric development. The stakeholders recognize the need to extend conservation efforts beyond protected areas and move toward coexistence with agriculture communities for the survival of the wild elephants. We mapped previous Human-Elephant Conflict (HEC) management methods and other governmental policies in Malaysia against the problem tree, and provided an overview of the different groups of stakeholders. The ToC was developed and adapted for each entity, while including Asian elephants as a stakeholder in the project. From the SROI estimation, we extrapolated the intrinsic value of the wild Asian elephant population in Johor, Malaysia, to be conservatively worth at least MYR 7.3 million (USD 1.8 million) per year. From the overall calculations, the potential SROI value of the project is 18.96 within 5 years, meaning for every ringgit invested in the project, it generates MYR 18.96 (USD 4.74) worth of social return value. There are caveats with using these value estimations outside of the SROI

context, which was thoroughly discussed. The SROI provides projects with the ability to justify to funders the social return values of its activities, which we have adapted to include the intrinsic value of an endangered megafauna. Moreover, SROI encourages projects to consider unintended impacts (i.e., replacement, displacement, and deadweight), and acknowledge contributions from stakeholders. The development of the problem tree and ToC *via* SROI approach, can help in clarifying priorities and encourage thinking out of the box. For this case study, we presented the thinking process, full framework and provided evidences to support the Theory of Change.

Keywords: human-elephant conflict, coexistence, theory of change, social return of investment, Asian elephant, *Elephas maximus*, Malaysia

INTRODUCTION

Southeast Asia (SEA) is a region rich in biodiversity with complex biogeographic divides (Hughes, 2017), with four subspecies of Asian elephants (one extinct), five subspecies of tigers (two extinct), three extant species of orang-utans, a marine region that is high in coral diversity and many more. Three out of 11 countries in SEA, including Malaysia, are recognized as megadiverse countries (von Rintelen et al., 2017). This region has a very high number of megafauna species facing the potential threat of extinction (Ripple et al., 2017), even though these megafauna are often regarded as charismatic species that attract public attention. One megafauna of concern is the Asian elephant (*Elephas maximus*), currently listed as "Endangered" on the IUCN Red List of Threatened Species (IUCN, 2019). Malaysia's Asian elephant subspecies include the Mainland Asian elephant (*Elephas maximus indicus*) in Peninsular Malaysia and the Bornean Pygmy elephant (*Elephas maximus borneensis*) in Sabah. Previously, in the eighteenth century, it was suggested that the elephant population in Peninsular Malaysia was a distinct subspecies, *Elephas maximus hirsutus*, described solely based on morphology description of a single baby elephant (Lydekker, 1914); however, term is not widely used. The Asian elephant is facing diverse threats throughout its range that include habitat loss and fragmentation, human-elephant conflict, and poaching (Sukumar, 2003; Fernando and Pastorini, 2011; IUCN, 2019; Mahmood et al., 2021).

One of the main challenges for wildlife research and conservation projects is in attracting long-term funders, as stakes are often high and with real possibilities of failures. Moreover, the scarcity of funds and a plethora of environmental and biodiversity related organizations, often results in high competition for project grants. Project planning tools such as problem tree and Theory of Change (ToC), have the potential to help projects strategize their approach and focus to create the most impactful change. It is useful in identifying suitable project Monitoring & Evaluation (M&E) indicators to monitor project progress (Rice et al., 2020). Donors themselves are very concerned about project outcomes and many funders are using M&E tools since the 1990s to monitor and measure project impact on the ground (Stem et al., 2005; Cameron, 2012). Key challenges in using M&E tools include the need to transfer project planning skills from the realm of expert planners to project executants (Cameron, 2012; Golini et al., 2018), and to define the real impact of the project on the ground (Stem et al., 2005). In terms of impact, it is often challenging to capture both the visible and invisible outcomes of the project in a quantifiable manner.

The Social Return Of Investment (SROI) approach is often used for measuring the social, environmental and economic impact of the social entrepreneurs, and it is usually conducted as a forecast at the conceptualization of the project or as an evaluation of the project after completion (Lingane and Olsen, 2004; Nicholls et al., 2021). What makes SROI unique from other project planning tools is the consideration given to both bad (usually unintended) and good consequences of the project. Based on the framework set by Nicholls et al. (2021), the SROI calculations include consideration if the project is taking over an existing activity that is producing the same change at the study site or if the project is moving the problem elsewhere (displacement). It includes the null hypothesis scenario, whereby if the project did not take place, would the project outcome still be realized (deadweight). Additionally, it requires the project proponent to give credit and acknowledgment to other players in the landscape (attribution). With this, SROI is able to guide the project executants to consider, in a holistic manner, the impact that they can create *via* the project and provide a transparent projection of social return value to the donors. There are concerns if SROI may be biased toward the "economic return" of investment, and conservationists may be wary that the measurement of "social and environmental values" in SROI may encourage "monetization" of the values. It is important to emphasize that the purpose of SROI is to help project executants to visualize the impact of the project on the ground for project planning and monitoring purposes, and additionally to provide justification to funders. The calculations for SROI cannot be used outside the scope of these purposes, and important caveats are further elaborated in the Discussion section. The framework deploy by SROI is to provide a centralized measurement for both tangible and intangible outcomes, mainly to help support management decisions.

In this study, we explore the use of the SROI framework (Nicholls et al., 2021) to support the development of Theory of Change for a human-elephant coexistence project *via* a collaborative approach with stakeholders. The Management & Ecology of Malaysian Elephants (MEME) is a project established in 2011 to conduct science-based research in

order to support evidence-based management of wild Asian elephants in Peninsular Malaysia. The project is carried out in collaboration with the Department of Wildlife and National Parks (PERHILITAN) in Peninsular Malaysia and with various other partners from non-governmental organizations, academia, and private sectors.

METHODS

Identifying Theory of Change and Human-Elephant Conflict Publication Trends

We examined the publication trends *via* the Web of Science search engine on 10th March 2021. The key phrase "Theory of Change," was searched through all the years with fixed word order, and subsequently the results were filtered with "Conservation OR Wildlife OR Environmental" keywords individually and collectively to examine the use of the theory in these fields. We examined the trend of Social Return of Investment *via* the more popular keyword "SROI" in combination with "Social" to avoid picking up other research or terms with identical acronyms. We used the keywords "wildlife" AND "conflict or coexistence," which generated more hits compared to "human-wildlife conflict," and filter the results for Asian elephants in general and specifically within South East Asian countries.

Constructing a Problem Tree

A problem tree is used for identifying issues or obstacles to the goal and to prioritize the issues along a cause-and-effect continuum (Alvarez et al., 2010). This exercise is conducted *via* a respectful discourse with stakeholders. Two planning exercises were conducted on 19th July 2018 and 7th November 2018 at the University of Nottingham Malaysia campus in Semenyih with the same group of 15 people attending both sessions. The discussion team includes a social scientist from the Nottingham Business School, two members of the IUCN Asian Elephant Specialists Group, two PERHILITAN staff who were managing human-elephant conflict cases on the ground and several other MEME researchers and students, with ages ranging from 20's to 50's. Collectively, the group represents more than 90 years of working experience (range: 3–20 years/ individual) from academic, non-governmental organizations, governmental agency, and the private sector (plantations and consultancies).

The group carried out *ad libitum* brainstorming to identify challenges to elephant conservation in Malaysia, represented by keywords written on flashcards. This was followed by the creation of the problem tree by the rearrangement of the flashcards along a cause-and-effect continuum, with the root cause at the bottom of the tree, and the effects placed upwards in the order of one (cause) leading to the other (effect) forming the branches of the tree. The process was moderated by the lead author who has prior experience conducting such planning exercises. Subsequently, the interrelationships between the issues were defined further using systems thinking (Haraldsson, 2004), whereby arrows representing same relationship or oppositional relationship were drawn to connect the issues. Two issues connected *via* arrows of same relationship (i.e., A increase B, and B increase A) then, it is considered a reinforcing loop. While two issues connected with opposing relationship (i.e., A increase B, but B reduces A) then it is considered as a balancing loop or negative-feedback loop (Mahajan et al., 2019).

Creating the Theory of Change

Theory of Change is a logical argument outlining the steps required to reach the goals and is recognized to be useful for tackling conservation conflicts (Baynham-Herd et al., 2018) and in helping conservation projects create impact (Stem et al., 2005). We develop the Theory of Change for individual stakeholders, using an emerging concept, SROI, to integrate social, economic and environmental values and quantify invisible outcomes of the project. The SROI mirrors the more popular project planning tool, Logical Framework Assessment, but with additional components. The concept of SROI is largely based on seven fundamental principles as quoted here: "involve stakeholders, understand what changes, value things that matter, only include what is material, do not over-claim, be transparent and verify the result" (Nicholls et al., 2021).

To carry out an SROI analysis, there are six stages or steps: "(i) Establishing a scope and identifying key stakeholders, (ii) Mapping outcomes, (iii) Evidencing outcomes and giving them a value, (iv) Establishing impact, (v) Calculating the SROI, and (vi) Reporting, using and embedding." An Excel template is used for capturing all critical information in a systematic manner and to derive the SROI ratio calculation (Nicholls et al., 2021).

Evidence for Theory of Change Based on SROI Approach

The foundation of SROI is the acknowledgment that project activities will actively create and/or destroy values, and result in changes (Lingane and Olsen, 2004; Nicholls et al., 2021). To measure changes that occur requires the project proponent to estimate a monetary value for the outcomes, which in turn need to be supported by evidence. The evidence provided is not expected to be accurate (approximation is sufficient), but it needs to be reliable, realistic and consistent. Based on Nicholls et al. (2021), the formula given for **Impact** value is denoted as the **Outcome** value after the deduction of deadweight, displacement, and attribution estimation. And the **SROI ratio** value would be the total impact value divided by total input.

$$\textit{Impact value} = \textit{Outcome} - \textit{Deadweight} - \textit{Displacement} - \textit{Attribution}$$
$$\textit{SROI} = \textit{Total Impact Value/ Total input}$$

To calculate **Input** value, in addition to funds given by donors, additional in-kind contributions by stakeholders such as direct participation in activity and sponsorship are included. We quantify the direct participation of stakeholders in activities in terms of hours or man-days for the whole project duration, which is then converted into **manpower value** by estimating the daily cost of hiring a daily paid assistant to do the work. Meanwhile, we

calculate the intrinsic value of **Outcome** by multiplying in-kind contribution (**manpower value**) invested by the stakeholder/s by the number of people in the community who will benefit from the investment. For example, an officer entrusted by an estate to learn and carry out safety guidelines for managing conflict with elephants may invest 1 day per week toward this purpose, but his or her action and knowledge could potentially benefit the safety of all staff and families staying in the estate. The SROI framework considers as well if the impact from the activity may last more than a year, and flexible enough for adjustment of success rates to account for some participants dropping out half-way or discontinuing the program after it ends.

We estimated the monetary value for wild elephants by extrapolating the results of a published study by Poh and Mohd Shahwahid (2008) that evaluated the average willingness to pay for wild elephant conservation and well-being as MYR 5.86/ person ($N = 200$) that was gathered from communities living in Human Elephant Conflict (HEC) area around Pahang, Terengganu and Taman Negara National Park. This value is potentially biased toward a lower value, as members of the public in urban areas have a higher appreciation for wildlife conservation (Guérin et al., 2017; Tan et al., 2020). We projected the value to a population of 32.6 million people in Malaysia (Department of Statistics Malaysia Official Portal, 2020) and prorated the value with estimated number of elephants in different parts of Peninsular Malaysia (Saaban et al., 2011) and Sabah (Alfred et al., 2011). We assumed that there will be no drop off in the intrinsic value of the elephants in the subsequent years of the project.

Deadweight estimation, requires the consideration that if the project did not take place, would the outcome (if not fully, then at what percentage) still be realized by other stakeholders? We derive the percentage calculation by considering ongoing efforts by stakeholders in the study area and denote 50% if there are other stakeholders with an overlap in activities and 25% if they are working on elephant conservation in general (without overlap). **Displacement** value is the consideration of whether the project is taking over an existing activity that is producing the same change or if the problem is being shifted elsewhere. Since the project is engaging with all parties who are actively working on wild elephant conservation at the study site as research partners, to build on each other's effort (avoid duplication) and jointly deliver the outcome, hence displacement is valued at 0%. The contributions from partners are captured under **Attribution**, which is the estimation of the efforts contributed by partners to help make the activity or goal successful. For activities that require a partner to participate fully (as part of empowerment), we denote the attribution value as 50%, and for activities that require direct support by partner/s to realize the outcome, we divided the percentage with the number of key sectors (i.e., government, public, private, and NGOs) involved. We are unable to outline fully each calculation and evidence prepared for ToC here, please see the full SROI framework under **Supplementary Table 1**.

RESULTS

Identifying Theory of Change and Human-Elephant Conflict Publication Trends

A general search of "Theory of Change" on the Web of Science (WoS) revealed 1,018 publications predominantly in the field of occupational health, education, psychology and social sciences from the last 20 years. Collectively, there were 102 ToC publications in conservation, wildlife and environmental fields with most articles being published in the last 6 years. Out of all, only nine publications found were on wildlife. Meanwhile, the search for SROI revealed only 145 publications, mainly in business economics, environmental sciences and social sciences published in recent years, but none for wildlife.

Although "wildlife" and "conflict OR coexistence" by themselves generated 4,127 publications on WoS, only 2.6% were from South East Asia, while 3.8% were on Asian elephants. Although there are ToC papers on poaching and wildlife trade, there are no ToC or SROI specifically for elephants.

Constructing Problem Tree and Identifying Objectives

The problem tree was constructed (**Figure 1**) by arranging challenges related to elephant conservation on a continuum scale with the causes at the "root" ascending to effects in the tree branches. The background for the challenges was elaborated in **Table 1**, and previous HEC mitigation in the past and other relevant efforts were captured according to the issues. Considering project limitations, we scope the project toward interventions targeting root, middle and top of the problem tree, focusing on "changing mindsets," "working with plantations to improve forest connectivity" and "fostering tolerance" respectively (**Table 2**).

Estimating Values and Evaluating Evidence for SROI

We calculated an intrinsic value of wild elephants for the study site in Johor to be at least MYR 7.3 million (∼USD 1.8 million) per year based solely on willingness to pay for the well-being of elephants in the forest, without considering elephants' ecosystem services or its role as an umbrella species that help conserve other wildlife. This intrinsic value of conserving wild elephants is shared equally with all key stakeholders as all sectors have to play a role to secure the existence of wild elephants in the landscape (see 2.4 Attribution).

The **total input** is estimated at the value of MYR 3.92 million for the period of 3 years. The **total impact value** was estimated to be at least MYR 14.59 million, with the SROI ratio of 3.72 (for every ringgit invested in the project, it brings a social return of investment worth MYR 3.72). When the project impact is projected for 5 years, the SROI per amount invested is 18.96.

We included Asian elephants in the study site as a stakeholder, alongside government agencies, non-governmental organizations, agriculture communities and

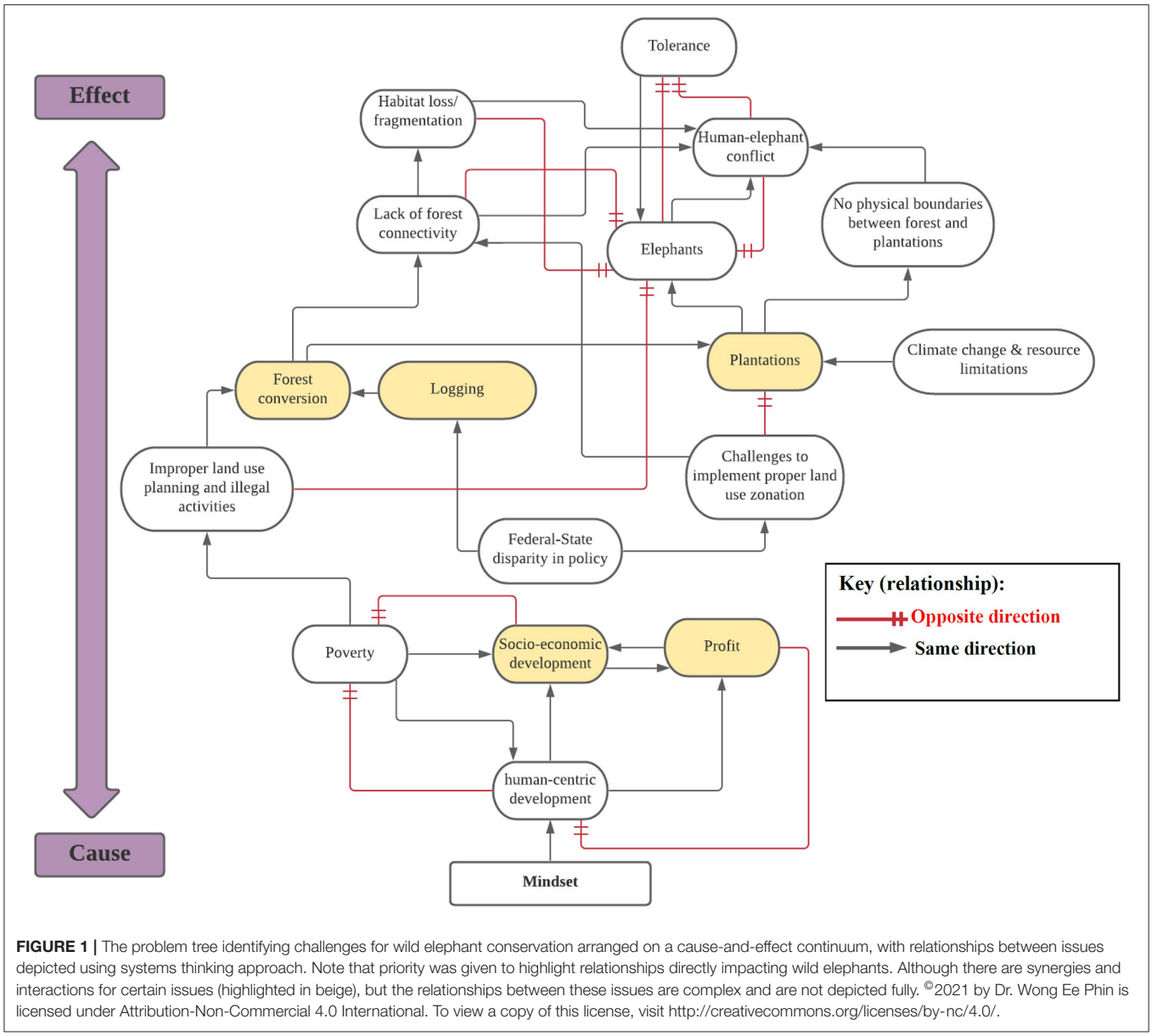


FIGURE 1 | The problem tree identifying challenges for wild elephant conservation arranged on a cause-and-effect continuum, with relationships between issues depicted using systems thinking approach. Note that priority was given to highlight relationships directly impacting wild elephants. Although there are synergies and interactions for certain issues (highlighted in beige), but the relationships between these issues are complex and are not depicted fully. ©2021 by Dr. Wong Ee Phin is licensed under Attribution-Non-Commercial 4.0 International. To view a copy of this license, visit http://creativecommons.org/licenses/by-nc/4.0/.

private sectors (**Table 2**), and justified the ToC and indicators of monitoring (see **Supplementary Table 1**).

DISCUSSION

The project Management & Ecology of Malaysian Elephants (MEME) used project planning tools such as problem tree and Theory of Change (ToC) to conceptualize a new phase of conservation work for elephants. We documented the thinking process and introduced the use of the Social Return Of Investment (SROI), which mirrors the logistic framework approach in the development of ToC, but with additional considerations for quantifying intangible outcomes.

Through the problem tree exercise, we acknowledged that in Malaysia, the established mindset of the government and society is to prioritize people's welfare first, and the country's development plans in the past have mostly been human-centric (Nagulendran et al., 2016). After World War II ended in 1945, Malaysia's concern was on alleviating poverty. After more than six decades of independence, Malaysia has managed to reduce her poverty level to 3.8% in 2009 (Hatta and Ali, 2013), however many indigenous communities are still living below National hardcore poverty line (Saifullah et al., 2021). These communities often face crop depredation and other types of conflict with wild elephants, although most are still influence by their ancestor's culture that imbued respect for the elephants (Lim, 2018). By applying systems thinking on the problem tree, which helps to visualize the intricacies of interrelationships between factor (Mahajan et al., 2019), we recognized that with reduction of the poverty rate and as the larger society becomes more affluent

(with an increase in profit), there are opportunities to shift the society's focus on human-centric development toward balanced development that supports wildlife conservation (Guérin et al., 2017; Tan et al., 2020) or toward a more eco-centric mindset (Taylor et al., 2020). In other parts of the world, there are similar shifts in societal values moving toward support for biodiversity conservation (Manfredo et al., 2021).

Malaysia is a signatory to various international treaties including the Convention of Biological Diversity, Aichi Targets and we have adopted the United Nations Sustainability Development Goals which may influence the trajectory to move away from a human-centric mindset (Government of Malaysia, 2012). Internationally, due to the demand of consumers for products with sustainable certification, plantations are extending their Corporate Social Responsibility remit toward nature and wildlife (Quilter, 2019). We identify this as an opportunity to bring on-board the wider society to support wild elephant conservation in particular. Furthermore, our past studies have indicated that wild elephants will be attracted to the agricultural landscape (de la Torre et al., 2021) for food (Terborgh et al., 2018; Ong, 2021), and simultaneously we recognized that plantations can potentially help to reconnect forest patches by establishing wildlife corridors (Department of Town Country Planning, 2009). By carrying out interventions that can help increase the tolerance of the agriculture communities toward wild elephants, and by reconnecting some of the larger forest patches, it may help to create more favorable circumstances to support the wild elephant population (**Figure 1**).

The objectives for the project are selected by considering the relationship between issues on the problem tree and the scope of the project. The prioritization of issues according to the cause-and-effect continuum effectively mean that interventions targeting issues closer to the roots will benefit issues above, as in efforts in tackling the cause can help minimize the effect. By designing a ToC that consider the relationship between issues, and by evaluating and prioritizing stakeholders together with SROI value, conservation projects, especially those dealing with conservation conflict, can identify areas where they can deliver the highest impact (Biggs et al., 2017; Rice et al., 2020). However this approach does not take into account the potential underlying conflict between stakeholders and hence further scoping work may be required on the ground to understand the power dynamics of the community (Zimmermann et al., 2020). During the course of project implementation, there could be a need to reiterate some parts of the planning process together with stakeholders on the ground (i.e., plantations and smallholders) to identify new issues and to verify assumptions. The challenge with project planning tools often occurs when the monitoring process becomes too rigid, which can impede the organic flow of the project's implementation on the ground (Stem et al., 2005; Cameron, 2012). We recommending projects to keep some flexibility in how activities can be carried out, considering planning is done at the conceptualization of the project often with general assumptions, while the reality on the ground could differ. We recommend projects to support their ToC assumptions with evidence and choose their indicators carefully to monitor the change that they want to see. Although SROI framework does not require accurate estimation of monetary values as long it is realistic and consistent, but often the danger is when these data are taken out of SROI context. Other common difficulties in implementing ToC include governance challenges, when the actual output depends on action from stakeholders who are higher up in the management hierarchy, or if the issue is extremely complex and requires a huge amount of effort in order to make a net benefit (Stem et al., 2005; Cameron, 2012; Biggs et al., 2017).

In this case study, the ToC was developed for stakeholders with the inclusion of Asian elephants as stakeholders in the SROI framework. We found the use of SROI can potentially account for invisible values, which can be further developed for wildlife projects by realizing that the society has an intrinsic appreciation of wildlife existence and there are social values when working together with stakeholders (Lingane and Olsen, 2004; Stem et al., 2005; Nicholls et al., 2021). The challenge would be to convert those values in monetary terms. Here, we calculated the monetary value for "in-kind contribution" by stakeholders *via* their time involvement with the project and extrapolated results from a "willingness to pay" study to quantify the intrinsic value of elephants. There is plenty of room for developing value quantification of ecosystem services provided by elephants, or in having tolerance toward elephants and many more.

The SROI calculations can help projects to reconsider the impact that they are making on the ground, and serve as a basis to justify to donors that the funds invested in the project is worth the social outcomes. However, there are some important caveats to consider, it is generally not encouraged to compare one project with another based on SROI ratios. The SROI is meant to help in monitoring internal progress or changes of the project from time to time. To interpret the SROI ratio for each project, we have to consider the local context, supporting evidence and the overall analysis of what factors that are being compared.

This case study calculated the SROI values for conserving an estimated 135 elephants in the State of Johor. Previously, Saaban et al. (2020) had used population viability analysis to predict that local extinction could happen to this elephant population if artificial removal from the wild continued. Using the intrinsic value of a wild elephant, extrapolated from a study on willingness to pay for a wild elephant's conservation and well-being, the SROI value generated was at least MYR 7.3 million/year for the elephant population in Johor. This is a very conservative estimation, and the intrinsic value calculated for wild elephants could potentially increase as more efforts are poured into conserving the species, higher awareness raised or when additional values such as elephant functions in ecosystem services are accounted for. This SROI value cannot be claimed by the project solely as it requires the involvement of all stakeholders including government agencies, non-governmental organizations, private sectors and communities to play vital roles in ensuring the survival and viability of the elephant population. Hence, the SROI template allows the project proponent to acknowledge the contributions from other stakeholders and present a realistic and transparent assessment to the funder.

TABLE 1 | Background to the issues presented in the problem tree, with past conservation efforts in Malaysia.

No.	Problem tree	Background	Past elephant related conservation efforts
1.	Tolerance	Farmers are unwilling to live alongside elephants (Ponnusamy et al., 2016). If the elephant population increases that could lower tolerance, but higher tolerance can allow more elephants to persist in the area. There is a taxonomic bias against conflict-prone megafauna and communities are calling for translocation of wild elephants (Tan et al., 2020). However, there is a positive trend among the general public that shows increasing concerns for wildlife conservation (Guérin et al., 2017).	Education and awareness program by PERHILITAN, MEME, and NGOs.
2.	Human-elephant conflict (HEC)	Elephants predating on crops resulting in economic losses, and creates conflict with farmers. In between 1998 and 2010, there were 10,759 HEC complaints with crop raiding being the majority of complaints (72.8%) (Saaban et al., 2011). Stakeholders affected by HEC consist of rubber and oil palm plantations (39.5%), smallholders (33.2%), villages (17.5%), and others (Saaban et al., 2011). Translocation is used to move elephants from conflict areas to contiguous forest areas. However, translocation is not a long-term solution (Wadey, 2020) and may have chronic effects on the elephants' stress response (Wong, 2018; Wong et al., 2018).	Translocation, electric fences, guarding, and chasing by PERHILITAN. Plantations have invested in electric fences and elephant ditches since 1980's.
3.	No physical boundaries between forest and plantations	Elephants venture into plantations and agriculture areas easily and this results in conflict (de la Torre et al., 2021). Electric fences and elephant ditches are very costly to build and maintain (Saaban et al., 2011; Ponnusamy et al., 2016).	Construction of electric fences, elephant ditches by PERHILITAN and plantations.
4.	Elephants	In the wild, elephants selectively choose palms, monocots and early succession plants to feed on (Terborgh et al., 2018; Ong, 2021). Elephants are attracted to agriculture land due to availability of food resulting in HEC. Agriculture land is a prime habitat for elephants (de la Torre et al., 2021).	Keep elephants out of agriculture land. Restrict elephant presences to protected areas and other contiguous forest.
5.	Habitat loss & fragmentation	There is a reduction in elephant range up to 68% in human occupied landscape when compared to 40 years ago (Tan, 2016). Roads can pose a barrier to elephant movement. The tendency for elephants to cross the East-West highway which cuts across two forest patches is decreased by 70% (Wadey et al., 2018). As forests shrink and are fragmented by linear infrastructure development, increasingly there is the need to reconnect forest patches *via* wildlife corridor traversing agriculture land (Department of Town Country Planning, 2009).	Implementation of the National Physical Plan and Central Forest Spine Master Plan.
6.	Lack of forest connectivity	The CFS Masterplan has identify important ecological corridors in Peninsular Malaysia (Department of Town Country Planning, 2009). The CFS corridors in the northern landscape are still connected for elephant use but are fragmented in the southern landscape of Peninsular Malaysia (de la Torre et al., 2019).	Implementation of the National Physical Plan and Central Forest Spine Master Plan. Johor Sustainability Policy 2017-2021 (Economic Planning Unit Johor, 2016).
7.	Climate change & resource limitation	Climate change will heighten the issue on food security. There are concerns over increasing competition over remaining available land, but at the same time protecting forests will be more important in increasing the country's resilience against climate change.	National Physical Plan (Department of Town Country Planning, 2016).
8.	Forest conversion	Some forest reserves are being converted to plantation forests or other land-use, which is a worrying trend (Miettinen et al., 2011; Law, 2020). Malaysia has pledged to retain 50% of land under forest cover in 1992 at Rio de Janeiro Earth Summit which helped to curtail some of the expansion (Varkkey et al., 2018).	Sustainability certification schemes UN Sustainable Development Goals
9.	Plantations (food security and economy)	Agriculture may not consider wildlife conservation as one of their main roles and place a bigger emphasis on importance in food security and contribution to the economy. Agriculture and infrastructure needs are often prioritized in land-use matters. Majority of damages to oil palm trees in plantations occur to palms aged 5 years and below (99%; Quilter, 2019). If plantation can concentrate HEC mitigation efforts in sensitive areas, and allow elephants to cross in matured oil palm areas—it could be possible to minimize the damage suffer by plantation and promote coexistence.	Sustainability certification schemes UN Sustainable Development Goals
10.	Federal-State disparity in policy	Dichotomy between Federal and State exists over governance of natural resources (Maniam and Singaravelloo, 2015).	State Parks and State sustainability plans. Eg., Johor Sustainability Policy 2017-2021 (Economic Planning Unit Johor, 2016).
11.	Improper land use and illegal activities	Illegal land clearing and poaching are major issues (Clements et al., 2010). Three groups of elephant poachers have been caught by PERHILITAN (pers. comm. Dr. Pazil bin Abdul Patah and En. Salman Saaban).	Joint wildlife enforcement efforts by PERHILITAN, Royal Malaysia Police, and the army. National Physical Plan (Department of Town Country Planning, 2016)

(Continued)

TABLE 1 | Continued

No.	Problem tree	Background	Past elephant related conservation efforts
12.	Challenges to implement proper land use zonation	The National Physical Plan (NPP) still requires States to adopt and implement Central Forest Spine ecological linkages. However, the Land Office often issue land tenures that clashes with NPP.	State-level sustainability plans. Eg., Johor Sustainability Policy 2017–2021(Economic Planning Unit Johor, 2016).
13.	Logging	State governments are still relying on logging for revenue (Law, 2020).	Case by case ecological fiscal transfer between the Federal government to State governments for not logging.
14.	Profit	Business may focus only on profits instead of contributing to the greater society and sustainability. Need more social entrepreneurs or non-profit/ conservation organizations and efforts (Abdul Kadir and Mhd Sarif, 2016).	Sustainability certifications UN Sustainable Development Goals
15.	Socio-economic development	Traditionally, emphasis is given to socio-economic development to reduce poverty. Thus, priority is often not given to biodiversity conservation.	Malaysia Plan (Rancangan Malaysia) Sustainability certifications UN Sustainable Development Goals
16.	Poverty	The HEC faced by the oil palm plantations occurred since the nineteenth century when Malaysia introduced a rural settlement scheme to reduce poverty and to promote national economic growth (Ahmad Zafir and Magintan, 2016). Poverty rate has since reduced (Hatta and Ali, 2013) but is still prevalent among the Indigenous communities (Saifullah et al., 2021).	Malaysia Plan (Rancangan Malaysia) UN Sustainable Development Goals
17.	Mindset—human centric development	Malaysia Plan (*Rancangan Malaysia*) prioritized the people's needs first. There is a need to mainstream biodiversity conservation and evoke a change in mindset among non-traditional conservation stakeholders to prioritize nature conservation as part of Malaysia's journey toward development.* **This planning exercise was conducted before Covid-19. As we move toward post-Covid 19, there is an urgency to create this shift in mindset.*	Mainstreaming of biodiversity *via* the National Policy on Biological Diversity 2016–2025 and Aichi Targets. National Elephant Conservation Action Plan UN Sustainable Development Goals Intergovernmental Panel on Biodiversity and Ecosystem Services Asean Biodiversity Center

We duly acknowledge that the intrinsic value of a wild elephant calculated here is purely an academic exercise and that the existence of any endangered species individuals is deemed priceless by the conservation communities (Soule, 1985) and any concerned citizen. However, in an effort to move the larger society toward supporting biodiversity conservation, increasingly the language of economics is used to justify the need for conservation despite challenges in capturing the complex relationship between nature and people *via* invisible and intrinsic values in addition to direct and indirect economic benefits (Kareiva and Marvier, 2012; Dasgupta, 2021). Instead of trying to force biodiversity calculations into traditional economic methods, the "Dasgupta Review" highlighted the potential to expand the ability of economic tools to take into account the holistic roles of biodiversity and nature, and their relationship with people (Dasgupta, 2021). But, until the world has fully embraced accounting of ecological footprint and biosphere regeneration (Dasgupta, 2021), we highly recommend our readers to avoid using the intrinsic economic value calculated for elephants outside of SROI context, as there are multiple assumptions used in the calculations and it may wrongly encourage the direct use of cost-benefit analysis to justify development above species survival (Catlin et al., 2013).

We like to emphasize that the real value of the development of ToC through the SROI approach is the ability to value the social (and biodiversity) returns of the conservation project itself, akin to social entrepreneurship, to justify to the funder of the project's necessity (Nicholls et al., 2021). Furthermore, the thinking process that the tools necessitate can help encourage the project proponents to consider thoroughly the value of change they may influence on the ground. The problem tree and SROI framework encourages collaboration with stakeholders to tackle critical issues (Rice et al., 2020), and consider both the positive and negative impact of the project carefully through the inclusion of replacement, displacement and deadweight calculations (Nicholls et al., 2021). By using the SROI framework to monitor the project development, projects can adjust their strategies based on adaptive management and make changes as the project goes along. Project management is often challenging due to the many moving parts and factors often outside of the project executants' control. The ToC and SROI system recommended here are approaches to help visualize the project challenges in a simplified and logical order, to support the design of interventions. The assumptions taken to design the interventions are often crucial, and often reiterations of the planning process (or some parts of it) may be needed at different

TABLE 2 | Stakeholders, theory of change, and possible indicators.

Stakeholders	Theory of change	Possible indicators	References
Asian elephants	1. Increase in wildlife-friendly practices in smallholder estates and plantations (i.e., setting aside movement corridors, removing snares, and stopping poachers)	Ability and willingness for the agriculture sector to adopt wildlife-friendly recommendations and guidelines.	Sustainability certifications United Nations Sustainable Development Goals
	2. Increase people's tolerance and willingness to live alongside elephants	Number or % of people willing to live alongside elephants increased. Based on comparison of baseline data and after intervention data.	Ponnusamy et al., 2016; Tan, 2016; Lim, 2018; Tan et al., 2020.
	3. Increase in habitat connectivity	Ability for elephants to move from one large forest patch to another, that links with the Central Forest Spine Master Plan.	Department of Town Country Planning, 2009
	4. Reduce the need for translocation of elephants in conflict areas	Number or % of people calling for translocation reduces. Comparison of baseline data and after intervention data.	Ponnusamy et al., 2016; Tan, 2016; Lim, 2018; Tan et al., 2020.
	5. Increase protected areas for Asian elephants and other wildlife	Increase in protected areas *via* State governments' willingness to support forest and wildlife protection.	IUCN World Protected Areas and UN Sustainable Development Goals.
	6. Maintain or expand the range of wild Asian elephants in Peninsular Malaysia based on (Tan, 2016)	Stop or reverse local extinction trend. Comparison of wild elephant range with baseline set in Tan (2016).	Tan, 2016; Saaban et al., 2020
Indigenous villagers	1. Increase in villagers' well-being and empowering them to participate in conservation of wildlife and forest habitat.	Ability to minimize HEC and support their family in terms of providing food, healthcare, sustainable economic income, and purpose in life. Number or % of people willing to live alongside elephants increased. Based on comparison of baseline data and after intervention data.	Ponnusamy et al., 2016; Tan, 2016; Lim, 2018; Tan et al., 2020.
	2. Increase in tolerance and willingness to live alongside elephants	Number or % of people willing to live alongside elephants increased. Based on comparison of baseline data and after intervention data.	Ponnusamy et al., 2016; Tan, 2016; Lim, 2018; Tan et al., 2020.
Smallholders	1. Increase in wildlife-friendly practices on estates (i.e., setting aside movement corridors, removing snares, and stopping poachers)	Ability and willingness for smallholders to adopt wildlife-friendly recommendations and guidelines.	Sustainability certifications United Nations Sustainable Development Goals
	2. Increase in understanding of the importance of conserving elephant range.	Number or % of people calling for translocation as HEC mitigation method reduces. Comparison of baseline data and after intervention data.	Ponnusamy et al., 2016; Tan, 2016; Lim, 2018; Tan et al., 2020.
	3. Increase in tolerance and willingness to live alongside elephants	Ability to minimize and/or tolerate HEC losses. Number or % of people willing to live alongside elephants increased. Based on comparison of baseline data and after intervention data.	Ponnusamy et al., 2016; Tan, 2016; Lim, 2018; Tan et al., 2020.
Plantations	1. Increase in wildlife-friendly practices on estates (i.e., setting aside movement corridors, removing snares, and stopping poachers)	Ability and willingness for plantations to adopt wildlife-friendly recommendations and guidelines (i.e., setting aside wildlife corridors)	Sustainability certifications United Nations Sustainable Development Goals
	2. Increase in plantation intervention to support smallholders and villagers in tolerating or managing HEC.	Ability and willingness for plantations to extend wildlife-friendly recommendations and guidelines to surrounding smallholders and villages *via* active involvement.	Sustainability certifications United Nations Sustainable Development Goals
	3. Increase in understanding of the importance of conserving elephant range.	Number or % of people calling for translocation as HEC mitigation method reduces. Comparison of baseline data and after intervention data.	Sustainability certifications United Nations Sustainable Development Goals
	4. Increase in tolerance and willingness to live alongside elephants	Ability to minimize and/or tolerate HEC losses. Number or % of people willing to live alongside elephants increased. Based on comparison of baseline data and after intervention data.	Sustainability certifications United Nations Sustainable Development Goals

(Continued)

TABLE 2 | Continued

Stakeholders	Theory of change	Possible indicators	References
Government (State and Federal)	Increase in support and implementation of policies to encourage the agriculture sector in adopting zero deforestation policies, restore degraded land, protect forest cover at key biodiversity and environment sensitive area (setting aside wildlife corridors and riverine buffer zone), implement best agriculture practices to conserve and protect rare, endangered and threatened species and secure the Central Forest Spine Master Plan.	Ability and willingness of the governmental agencies to support the project.	National Elephant Conservation Action Plan National Policy on Biological Diversity 2016–2025 and Aichi Targets. UN Sustainable Development Goals
Project Staff	1. Increase in capacity building, personal development and project management	Ability to facilitate and manage projects and build relationships with stakeholders.	Quarterly report. Annual report
	2. Increase in scientific knowledge	Ability to produce scientific papers and policy recommendations	Quarterly report, Annual report

management levels, with different groups of stakeholders, or at different phases of the project to identify new issues and help verify assumptions.

CONCLUSION

Project planning tools can help wildlife conservation projects in prioritizing issues to tackle and stakeholders to engage with, in order to achieve its objectives. However, the true value is in the process of deliberation and constructive discussion, which allows thinking out of the box, and building cooperation between stakeholders. Tools like ToC and SROI can provide further justification to donors and convince them on the potential project outcome. We recommend projects to have some flexibility in envisioning and carrying out activities on the ground and to select their project indicators carefully.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

ETHICS STATEMENT

Ethical review and approval was not required for this study as it is not a social survey but a project planning exercise involving all coauthors in this paper.

AUTHOR CONTRIBUTIONS

EPW facilitated and moderated the project planning processes and conceptualized and took the leading role in writing this paper. ACA, NZ, PC, AGQ, JAT, ASM, WHT, LO, MAR, SSD, VP, TWL, OOC, AFA, NH, AT, JW, VL, AZ, MMI, PAP, MTAR, and SS have participated in the planning process or contributed own research, shared feedback, and contributed to this write-up. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.682590/full#supplementary-material

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The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Examining Drivers of Divergence in Recorded and Perceived Human-Carnivore Conflict Hotspots by Integrating Participatory and Ecological Data

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Human-carnivore conflict is a global challenge with complex and context-specific causes and consequences. While spatial analyses can use ecological principles to predict patterns of conflict, solutions to mitigate conflict must also be locally adaptable, sustainable, and culturally-sensitive. In Nakuru County, Kenya, rapid development and land subdivision have exacerbated conflict by isolating wildlife in protected areas that are increasingly adjacent to human settlements. In an effort to understand local perspectives on carnivore conflict, and to apply this information toward locally-based conservations actions, we conducted gender-stratified interviews and participatory mapping sessions with 378 people in 16 villages near two ecologically isolated protected areas in Kenya: Lake Nakuru National Park and Soysambu Conservancy. Specifically, we developed a method for associating interview responses and demographic information with spatial participatory data to examine how local perceptions of conflict compared to spatially-explicit records of livestock depredation in the region from 2010 to 2018. We mapped kernel densities of recorded and perceived risk of human-carnivore conflict and then tested for potential social and ecological predictors of divergences found between the two datasets. Mismatched hotspots of observed and perceived risk of conflict were correlated with several ecological and socioeconomic factors. Regions with higher NDVI exhibited more perceived conflict, while the opposite held true for verified conflict. Road density was positively correlated with both types of conflict, and both types of conflict increased closer to protected areas. Livestock ownership, visitation to Lake Nakuru National Park, if the participant's child walked to school, and male gender identity were associated with more perceived conflict reports. Education level and national park visitation were associated with more positive attitudes toward carnivores. Our results show that while observed and perceived conflict may ultimately be equally important for understanding and managing human-carnivore conflict, they may be driven by markedly different social and ecological processes. We suggest that integrating the spatially explicit experiences and perspectives of local communities with more traditional ecological methods is critical to identifying lasting and socially just forms of conflict mitigation.

Keywords: human-carnivore conflict, risk perceptions, conservation practice, participatory mapping, socioecological

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INTRODUCTION

Human-carnivore conflict (HCC) is a primary driver of large carnivore declines globally (Ripple et al., 2014) and creates a significant challenge to rural livelihoods in many areas (Muhly and Musiani, 2009). For example, in the United States, over $168 million in livestock losses per year are attributed to depredation by carnivores (USDA Animal and Plant Health Inspection Service, 2020). Livelihood impacts of HCC are most pronounced in regions where carnivore populations remain viable or have recovered, and where marginal incomes place producers near poverty (Dickman et al., 2011). Along with affecting livelihoods, human-wildlife conflict is known to have a number of indirect social and emotional impacts on affected communities, such as diminished psychological well-being and food insecurity (Barua et al., 2013). Conflicts between people and carnivores are exacerbated by a combination of sociopolitical factors (e.g., regional livelihoods, poverty, global wildlife policies; Treves and Karanth, 2003) and local histories of people's relationships with wildlife (e.g., Megaze et al., 2017), as well as increased development that has intensified habitat fragmentation and human-wildlife interactions (Were et al., 2013; Weldemichel and Lein, 2019). Thus, human-carnivore conflicts comprise impacts associated with interactions between carnivores and people, along with the human-human relationships underlying and influencing those interactions (Young et al., 2010; Redpath et al., 2015).

Conservation biologists lean heavily on an understanding of ecology when researching and managing human-carnivore conflict (Wilkinson et al., 2020). In recent years, there has been considerable momentum behind using ecological data in combination with innovative spatial tools for addressing conflict using scientific evidence (Miller, 2015; Miller and Schmitz, 2019). Predation risk mapping, for example, layers verified conflict events across ecological (e.g., habitat structure and productivity) and anthropogenic (e.g., human infrastructure and activity) variables in order to overcome HCC's inherent context-dependency, and to anticipate future carnivore conflict (e.g., Broekhuis et al., 2017). For instance, in arid ecosystems, conflict has been observed to increase in the rainy season when wildlife are not reliant on permanent water bodies and are able to disperse widely (Koziarski et al., 2016). Yet, in fenced arid ecosystems, wildlife transgressions of fences to exit protected areas may be higher in the dry season (Kesch et al., 2015), possibly because seasonal vegetation resources are more limited within fenced ecosystems than in unfenced ecosystems (Bartzke et al., 2018). Thus, predation on livestock and carnivore attacks on people in different regions with varying human development may exhibit measurable, context-specific, and spatially-explicit patterns across key ecological variables (Thorn et al., 2012). Additionally, anthropogenic structures and activity have altered wildlife behavior and ecology around the globe at numerous scales (Gaynor et al., 2018; McInturff et al., 2020), and may be consequential covariates when mapping carnivore conflicts with people. Risk mapping and other spatial methods have thus proven to be highly useful tools for quantifying correlates of verified conflict and employing ecological theory to create targeted mitigation strategies that address HCC (Melzheimer et al., 2020).

While global increases in HCC are regularly studied by examining the associations between ecological covariates and verified on-the-ground human-carnivore conflict reports, there is increasing understanding that the perception of risk held by local communities may more meaningfully predict their attitudes toward carnivores and their retaliatory or preventative actions (Dickman et al., 2014). Though interactions between wildlife and humans are situated within a broad range of social, institutional, and ecological landscapes, a key element of any human-carnivore interaction is human behavior (Lischka et al., 2018). Behavior of people when interacting with wildlife is, among other factors, driven by emotion, experience, and resulting attitudes and perceptions (Carter et al., 2012a), making human emotions and perceptions critical for understanding and resolving conflicts between people and carnivores.

A number of studies have acknowledged that perceptions of conflict can diverge from ecological findings and yet still provide tangible contributions to conservation efforts (Siex and Struhsaker, 1999; Dickman et al., 2014). Some of these have employed surveys to better understand the drivers of people's perceptions of conflict in space and time (e.g., Holmern et al., 2007). These studies and others suggest the most important observed social drivers of HCC perception, realization, and management outcomes among stakeholders are gender, education level, livestock ownership and adoption of tools for guarding livestock, and visitation and access to nearby protected areas (Tessema et al., 2010; Knopff et al., 2016; Mkonyi et al., 2017). For example, men and women may have different motivations, goals, and risk perceptions regarding human-wildlife conflict and management (Gore and Kahler, 2012), and women may bear disproportionate burdens of conflict due to gendered relations of space and identity (Ogra, 2008). Additionally, education level may influence attitudes toward wildlife and conservation (Akama et al., 1995; Holmern et al., 2007; Dressel et al., 2014; Megaze et al., 2017), and may also be an indicator of modernization, which is hypothesized to increase positive attitudes toward carnivore conservation (Bruskotter et al., 2017).

Livestock ownership also plays a potentially major role in perceptions of conflict since livestock owners are most likely to fear predation's impact on their livelihoods. These same stakeholders may be more likely to discuss conflict history or their perceptions of risk with neighbors (Kellert, 1985), which can contribute to spreading of perceived risks (Dickman et al., 2014). Relatedly, the adoption of common interventions designed to reduce carnivore conflicts (such as fladry, lights, noisemakers, etc.; van Eeden et al., 2018) may also impact people's perceptions of carnivores, conflict, and risk (Eklund et al., 2020). Number of livestock owned (Hemson et al., 2009) as well as number of children or family size (Khumalo and Yung, 2015), may also be indicators of financial precarity that influence conflict risk perceptions. Finally, national park visitation, as both a means of ecological education (e.g., Tomicevic et al., 2010) and connection to wildlife living on the landscape (e.g., through ecotourism; Waylen et al., 2009), may have the potential to affect

community members' understanding of and thus reaction to carnivores (Espinosa and Jacobson, 2012; Mkonyi et al., 2017). These social factors can be as critical as ecological variables when understanding and predicting patterns of HCC across different landscapes.

While these and other socioeconomic factors help predict local perceptions of conflict, the application of information on perceptions to structure and implement programs HCC is rare (Lozano et al., 2019). Moreover, the participatory methodologies necessary to assess and apply human perceptions are scarce across human-wildlife conflict research (Gray et al., 2020). This is despite the known importance of considering spatial, ecological, and social variables together for long-term conflict mitigation (White et al., 2009), and the common acknowledgment that conservation conflicts are best managed when science and solutions are co-created with affected communities (Treves et al., 2009; Redpath et al., 2013). In fact, examples abound of cases where a lack of participatory and integrative approaches have contributed to ineffective, short-lived, and/or unjust solutions to conflict (Meguro and Inoue, 2011; Eklund et al., 2020). For targeted and effective outreach and management of HCC, we need to address this disconnect by working toward an understanding of how and why verified and perceived HCCs diverge (Dickman, 2010), as well as how conflict risk perceptions cluster spatially and are driven by various social and ecological factors (Bruskotter and Wilson, 2014).

Here, we sought to bridge this gap by using a unique combination of verified conflict reports and participatory perception data to answer the following questions: (1) How do verified and locally perceived carnivore conflict compare spatially?, (2) How are similarities and differences in the two datasets correlated with ecological variables and infrastructure?, and (3) Are there social predictors (e.g., demographics, livestock ownership, and attitudes toward carnivores) of the level and distribution of perceived carnivore conflict? We examined these questions in the region surrounding Lake Nakuru National Park and Soysambu Conservancy in the Rift Valley of Kenya. This location provided an ideal system for this study because of its high rate of human immigration and land subdivision, and the resulting close proximity of wildlife to people, human activities, and infrastructure (Kassilly et al., 2008; Mubea and Menz, 2012; Wilkinson et al., 2021). We predicted (1) that verified and perceived conflict would exhibit observable spatial differences, and (2) that these disparities would be driven by a variety of ecological factors, such as season, vegetation, road density, and distance to protected areas, as well as social factors, such as participant education level, gender, livestock ownership and activities, and national park visitation (**Table 1**).

METHODS

Study Site

We conducted our study in Nakuru County, in the Rift Valley, southwest Kenya (**Figure 1**) from June 2018 to March 2019. The study area (∼500 km^2, 0°26' S, 36°1' E) includes two major wildlife protected areas: Lake Nakuru National Park (LNNP, 188 km^2), which is one of two fully fenced national parks in Kenya, and Soysambu Conservancy (190 km^2), which is semi-fenced and functions simultaneously as a wildlife conservancy and a livestock ranch with over 10,000 cattle, sheep, and goats. The two large alkaline lakes in the region, Lake Nakuru and Lake Elmenteita, are designated UNESCO World Heritage sites. The region supports many species of large mammals, including threatened and endangered species such as black rhinoceros (*Diceros bicornis*) and Rothschild's giraffe (*Giraffa camelopardalis rothschildi*); large carnivore species, such as African lion (*Panthera leo*), spotted hyena (*Crocuta crocuta*), and leopard (*Panthera pardus*); and several mesocarnivore species, such as serval (*Leptailurus serval*) and black-backed jackal (*Lupulella mesomelas*). Many carnivore populations in the region (both inside and outside of protected areas) are stable or increasing despite heavy historical persecution (Ogutu et al., 2017).

Outside of protected areas, the Nakuru-Elmenteita watershed is home to dense human populations, with considerable immigration into the region. Small-scale agriculture and pastoralism, as well as increased urbanization, are common in the settled areas surrounding LNNP and Soysambu Conservancy, and there is a mix of ethnic representation (mostly Kikuyu, Kalenjin, and Maasai). Nakuru town, which is directly adjacent to the northern border of LNNP, is home to an estimated 570,674 people (Kenya National Bureau of Statistics, 2019) and is considered to be one of the fastest growing cities in East Africa. In many places throughout the study area, human settlements directly about the conservancy and park boundaries.

Participatory Data

Data Collection

In order to gather community perspectives on carnivores and conflict, we selected 16 sub-villages, within 5 broader village areas, located within 5 km of the protected area boundaries (**Figure 1**). The sample comprised representation from every rural village with lands adjacent to the two protected areas, while excluding urban areas. Though participatory mapping is subject to inherent logistical, access, and scalability limitations (Brown, 2012), we addressed these challenges in a number of ways through iterative pre-testing and sampling considerations. Because the study area was large and many of the households were unmapped, we used semi-random heterogeneity sampling (Blankertz, 1998) to identify 378 participants (180 women, 198 men) for participatory mapping and interview sessions. Participants were informed (a) they could leave mapping and interview sessions at any time, (b) that participation in the exercise was not mandatory, and (c) that compensation was not provided. To reduce bias in responses, participants were informed that the interviewers and facilitators were students, and that the students held no direct authority in addressing human-wildlife conflict issues.

For each participatory mapping session, we aimed for 12 participants, with no more than 6 participants drawing on a single map (for legibility purposes). However, this wasn't always feasible, as the mapping sessions were popular and occasionally drew crowds. Thus, in a few circumstances, up to 8 participants drew on a single map. Mapping sessions were gender-stratified,

TABLE 1 | Hypotheses related to verified conflict, perceived conflict, and attitudes toward carnivores.

Variable	Hypotheses	References
Education level	Increasing education level is correlated with 1) more positive attitudes toward carnivores conservation, and 2) fewer perceived conflict reports.	Akama et al., 1995; Holmern et al., 2007; Dressel et al., 2014; Bruskotter et al., 2017; Megaze et al., 2017
National park visitation by participant or child	Participants who have visited the national park, or whose children have visited the national park are more likely to 1) have positive attitudes toward carnivores, and 2) report fewer perceived conflicts.	Tomicevic et al., 2010; Espinosa and Jacobson, 2012; Hausmann et al., 2016; Mkonyi et al., 2017
Active nighttime livestock guarding	Livestock owners who employ tools to actively guard their livestock at night are more likely to 1) have positive attitudes toward carnivores, and 2) report fewer perceived conflicts.	Holmern et al., 2007; Rust et al., 2013
Perceived threats to children	Participants who believe carnivores pose a threat to children on their way to school are more likely to 1) have negative attitudes toward carnivores, and 2) report more perceived conflicts.	Bruskotter and Wilson, 2014; Bruskotter et al., 2017
Number of children	Increasing number of children is correlated with 1) negative attitudes toward carnivores, and 2) more perceived conflict reports.	Khumalo and Yung, 2015; Bruskotter et al., 2017
Gender	1) Women are more likely to have negative attitudes toward carnivores. 2) Men are more likely to report more perceived conflicts.	Ogra, 2008; Gore and Kahler, 2012
Livestock ownership	1) Livestock owners are more likely to a. have negative attitudes toward carnivores, and b. to report more perceived conflicts. 2) These effects are stronger with increasing number of livestock owned.	Kellert, 1985; Hemson et al., 2009; Dorresteijn et al., 2014; Dressel et al., 2014
Season	1) Verified conflict increases during the rainy season. 2) Perceived conflict shows no difference between seasons.	Koziarski et al., 2016; Bartzke et al., 2018
Vegetation greenness (NDVI)	1) NDVI is positively correlated with both perceived and verified carnivore conflict.	Thorn et al., 2012; Koziarski et al., 2016; Bartzke et al., 2018; Ugarte et al., 2019
Road density (as a proxy for human population)	1) Road density is positively correlated with both perceived and verified conflict.	Treves and Karanth, 2003; Ugarte et al., 2019
Distance to protected area	1) Distance protected area is negatively correlated with both perceived and verified conflict, with a stronger effect for verified conflict.	Mkonyi et al., 2017; Ugarte et al., 2019; Weldemichel and Lein, 2019; Gray et al., 2020

with men and women gathering on different days to encourage open conversation and a broad range of perspectives (Pfeiffer and Butz, 2005). A total of 322 maps were drawn across all participatory mapping questions and sessions.

Each interviewee's session began with a short (∼10 min) one-on-one interview, with a Kenyan master's student serving as an interviewer, using the application Open Data Kit (ODK; https://getodk.org/), carried out on Android devices (Motorola Moto E). Interviews were conducted in English or Kiswahili, depending on the interviewee's preference. Information gathered included demographic data, risk perceptions about carnivores, attitudes toward carnivores and carnivore conservation, livestock ownership, experience with carnivore-livestock conflict, educational experience, national park visitation, and employment (**Appendix 1**), with a combination of multiple choice, check all that apply, numerical, and open-ended questions. Prior to the interview, each participant was assigned a unique pen color for the day. During the initial interview, a photo was taken of their pen within the ODK application. This allowed us to associate a participant's spatial data with their interview data while maintaining anonymity.

For the participatory mapping portion of the sessions, paper maps were developed and printed using *Field Papers* (www.FieldPapers.org). Field Papers is an open source tool to print basemaps that can be annotated in the field and then scanned, allowing annotation to be digitized into a GIS database. During the sessions, participants were first given a minimum of 15 min of map orientation, though these orientation exercises and conversations often lasted longer than 30 min. Participants were encouraged to teach one another by using laminated, highly detailed atlases of the region, and finding locations of interest to the community such as the national park, particular intersections, Nakuru town, and village centers. We asked participants to use the assigned pens and paper maps we provided to draw their answers to 24 general questions regarding places of importance, livestock predation, carnivore presence, desired carnivore conservation and movement, risk perceptions regarding carnivores, and other factors (**Appendix 1**). For each question on each map, a unique (to that map) symbol (falling into the categories of point, line, or polygon) was requested. Participants were encouraged to draw on top of one another's symbols as needed.

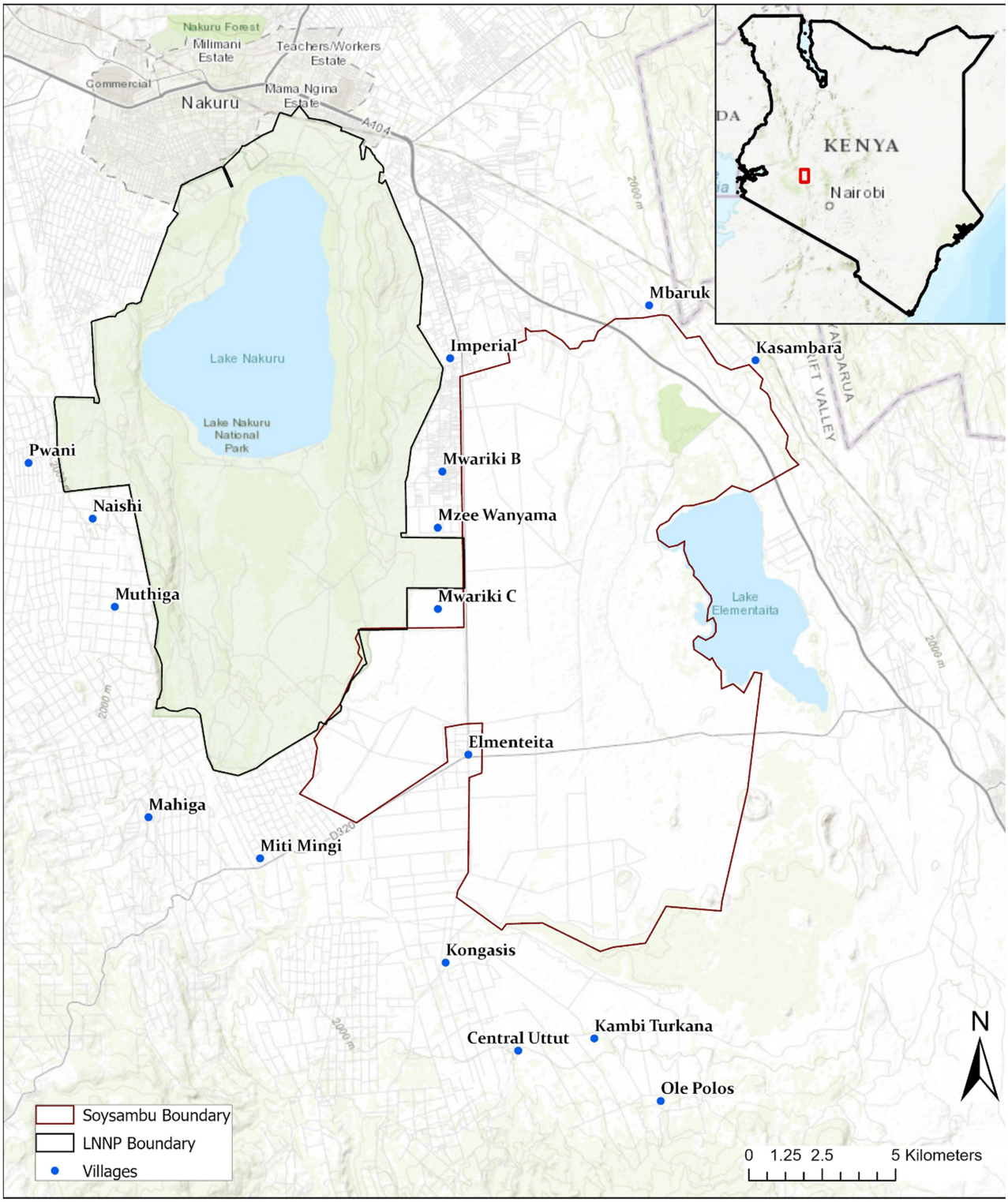


FIGURE 1 | Map of the study area, including villages surveyed.

Participatory Data Preparation

To digitize maps, participatory maps were photographed, and a QR code allowed the map images to be georeferenced directly using FieldPapers.org. Georeferenced map images were then uploaded into ArcGIS Pro (Version 2.5) in order to trace each question's spatial responses (points, lines or polygons) into GIS layers, with each layer representing the collected answers to one question. During the digitization process, pen colors on each session's maps were again cross-referenced with photographs of pen colors that had been automatically labeled with each interview's unique identification number. These identification numbers were assigned to each feature in each layer's attribute table. Interview data were then joined with attribute tables for each layer, and each question's layers were subsequently merged into a single master layer that included data from all sessions for that question.

Verified Conflict Data

Human-carnivore conflict (HCC) data for 2008–2018 were provided by the Kenya Wildlife Service. The dataset contains HCC incidents (such as carnivore attacks on livestock and threats to people) reported to the Nakuru Community Wildlife Service (CWS) station. The station houses trained rangers who respond to conflict issues and also undertake community outreach around the park and adjacent localities within the county. Conflict cases are reported by the local community through a dedicated telephone hotline or the institutional call center, both of which are open 24 h a day. Once a conflict report is received at the station, a CWS team is dispatched to verify. The details of the nature of each conflict are collected by the rangers and later recorded in an occurrence book. The information collected includes the date, location name, conflict species, the nature of the conflict, and the management action taken. Data recorded in the occurrence book are later entered into a database at the station. We obtained these verified conflict data from the main human-wildlife conflict database and georeferenced each record to the approximate village or sub-village level using landmarks and location names provided in the original dataset. While this dataset consisted of historical records of conflict which were initially collected solely for monitoring purposes (see Easterday et al., 2018), the data were cleaned and georeferenced points were iteratively verified with Kenya Wildlife Service staff prior to analysis.

Spatial Explanatory Variables for Conflict Reports

The ecological and anthropogenic spatial covariates that we tested as predictors of conflict reports included distance to protected area, road density (kernel density, per km^2), mean vegetation greenness (as measured by NDVI- normalized difference vegetation index, via Landsat 8, for 2018), and slope (via Shuttle Radar Topography Mission, 30 m).

Data Analysis

Summary and Comparison of Verified and Perceived Conflict

To determine spatial differences between the verified and perceived datasets, we used ArcGIS Pro to conduct kernel density estimations (KDE) for the entire verified conflict and perceived conflict datasets, respectively. From the KDE analyses (search radius = 3 km) we created difference maps comparing the verified and perceived datasets across all stratifications by subtracting the verified conflict KDE from the perceived conflict KDE. We thresholded difference maps to the upper and lower quantiles to determine areas of highest disagreement among the two datasets, and conducted generalized linear regressions to assess perceived and verified conflict density in relation to distance to protected area, road density, NDVI, and slope.

Correlates of Local Clustering of Perceived Conflict

In order to determine whether people with positive attitudes toward carnivores and who guard their livestock nonetheless exhibited significant clustering in their perceived carnivore conflict reports, we first used global logistic regressions to identify predictors (**Supplementary Table 1**) of (1) attitudes toward carnivores and (2) nighttime livestock guarding behavior. We then employed a geographically weighted logistic regression (GWLR; Brunsdon et al., 1996) to test for local clustering. The initial regressions revealed a best-supported model (AUC = 0.805) that included the following variables to retain for GWLR: cow ownership, sheep or goat (hereafter *shoat*) ownership, whether the participant collected water in the evening (i.e., landscape traversal at night), number of reasons reported for hyenas to be conserved, belief that hyenas have access to too few wild prey, and perceptions of carnivore-related threats to children on their way to school (**Table 2**).

Trends in Verified Conflict Reports

We used linear regression to test for trends in HCC reports over time for each carnivore species, for each livestock species and for humans, and for all carnivore species in aggregate using *R* (R Core Team, 2018). For all non-spatial analyses, verified conflict data from 2013 were excluded because reports were only recorded for 1 month of that year.

Predictors of Perceived Conflict/Risk and Attitudes Toward Carnivores

To determine correlates of perceptions, we assessed the correlates of two variables: perceived carnivore conflict and attitudes toward spotted hyenas. As a widely reviled carnivore species in sub-Saharan Africa (Glickman, 1995), and as one of the most populous and visible carnivores in this region (Wilkinson et al., 2021), spotted hyenas served as the best proxy for examining what drives differences in attitudes toward conflict-prone carnivores among Nakuru County residents. Thus, the main proxy for attitudes utilized in this study was "Do you think it is important to conserve the spotted hyena?" Pair-wise analyses were conducted for all relevant explanatory variables (age, education, national park visitation by participant, national park visitation by participant's child, livestock ownership, whether participant actively guards livestock at night, number of livestock owned). To determine whether perceived carnivore conflict or attitudes could be predicted using these variables, a logistic regression was then run for each dependent variable across all explanatory variables. After eliminating any collinear variables using the *vif* function, we used the *dredge* function in the *MuMin*

TABLE 2 | Variables retained in best-performing model of predictors of attitudes toward spotted hyena conservation (AUC = 0.805).

Variable	β	SE	*z*-value	*p*-value
Has experienced an attack on their cattle*	−1.758	0.945	−1.86	0.063
"Hyenas are attacking livestock due to food shortage*"	0.793	0.29	2.731	0.006
Time of day participant leaves their home for food	0.157	0.092	1.712	0.086
Fears carnivores on child's way to school*	0.68	0.288	2.358	0.018
Visited LNNP*	0.738	0.289	2.55	0.011
Primary occupation: farmer/herder	−0.682	0.302	−2.255	0.024

**Y/N questions: "No" is the reference variable.*

package in *R* to conduct model selection, and retained model variables within 2 delta AIC of the top model for model averaging (Burnham and Anderson, 2002). To test the robustness of the top model, we bootstrapped a calculation of the area under the receiver operating characteristics curve (AUC; Pearce and Ferrier, 2000). We randomly split the data into 20% testing and 80% training data, and calculated AUC using the *performance* function in the *ROCR* package. AUC values below 0.7 were considered poor, values between 0.7 and 0.8 were considered acceptable, and values >0.8 were considered good or excellent (Hosmer and Lemeshow, 2000).

Finally, to assess underlying values and beliefs that may lead to positive or negative attitudes toward carnivores, we conducted descriptive statistics and pairwise analyses of responses to follow-up questions in which we had asked people to describe why they did or did not believe spotted hyena conservation was important (see **Appendix 1**).

RESULTS

Spatial Patterns of Verified and Perceived Carnivore Conflict

Overall Patterns and Correlates

Verified and perceived conflict reports exhibited marked differences in spatial distribution and density. Kernel density estimates revealed a maximum of 3.34 verified and 3.44 perceived conflict reports per km^2 within the study area (**Figures 2A,B**). The difference map ($KDE_{perceived}$ - $KDE_{verfied}$) showed a maximum of 3.02, and a minimum of −3.44, with a mean difference of 0.044, meaning differences in the mapped reports across the study area skewed slightly toward perceived conflict. However, the minimum indicated a region on the map where there were no perceived conflict reports at all (**Figure 2C**).

KDE analyses exhibited 198.88 km^2 of high divergence (quantified as first [−3.44 to −0.444] and eighth [0.709–3.02] quantiles of difference) between the two datasets: 87.02 km^2 (∼9.8% of the KDE study extent) skewing toward perceived conflict, and 111.856 km^2 (∼12.6% of the study area) skewing toward verified conflict (**Figure 2A**). Within these areas of maximum divergence, mean NDVI for the lower quantile (i.e., areas skewed toward verified conflict) was 0.283 (σ = 0.038), while mean NDVI for areas skewed toward perceived conflict was 0.316 (σ = 0.015). NDVI was positively correlated with perceived conflict kernel density (β = 0.477, $p < 0.01$), and negatively correlated with verified conflict kernel density (β = −0.413, $p < 0.01$). Mean road density within areas of maximum divergence was 2.46 (σ = 1.113) for areas skewed toward perceived conflict, and 2.23 (σ = 0.341) for areas skewed toward verified conflict. Road density was positively correlated with both perceived (β = 0.115, $p < 0.001$) and verified (β = 0.103, $p < 0.001$) conflict density. Distance to protected area was strongly negatively correlated with perceived (β = −13.327, $p < 0.001$) and verified (β = −13.794, $p < 0.001$) conflict, while slope showed a slight negative correlation with both perceived (β = −0.009, $p < 0.05$) and verified (β = −0.016, $p < 0.001$) conflict.

Correlates of Local Clustering of Perceived Conflict

Those engaged in nighttime guarding of cattle did not differ from others in their spatial perceptions of HCC hotspots (**Figure 3A**), but geographically weighted logistic regression revealed local clusters (**Figures 3B,C**). GWLR results indicated that perceived carnivore-related threats to children and beliefs that wild prey was scarce correlated with local clusters of perceived conflict that were reported despite guarding behavior (**Figures 3B,C**).

Those who reported positive attitudes toward spotted hyenas similarly did not exhibit marked clustering in their perceptions of HCC hotspots. Similarly to the guarding behavior results, GWLR revealed that park visitation, perceptions of carnivore-related threats to children, and nighttime livestock guarding correlated with varying local clusters of perceived conflict reports in relation to attitudes (**Figure 4**).

Trends in Verified Conflict Reports

There was an upward, but non-significant, pattern in overall verified conflict reports over time (**Figure 5A**). However, carnivore species exhibited different trends over time: there was a slight downward trend in proportion of conflicts attributed to leopards (β = −0.0399, $p < 0.01$), an upward trend in the proportion of conflicts attributed to servals (β = 0.0189, $p < 0.01$), as well as non-significant upward trends in proportion of conflicts attributed to spotted hyena (β = 0.0091, p = 0.428) and lion (β = 0.0119, p = 0.331) (**Figure 5B**). As far as livestock attacked, verified conflict reports concerning sheep (β = −0.0145, $p < 0.05$), and dogs (β = −0.0129, $p < 0.05$) decreased over time (**Figure 5C**). A higher number of verified conflict reports were reported during the dry season, but this result was non-significant (**Supplementary Figure 1**).

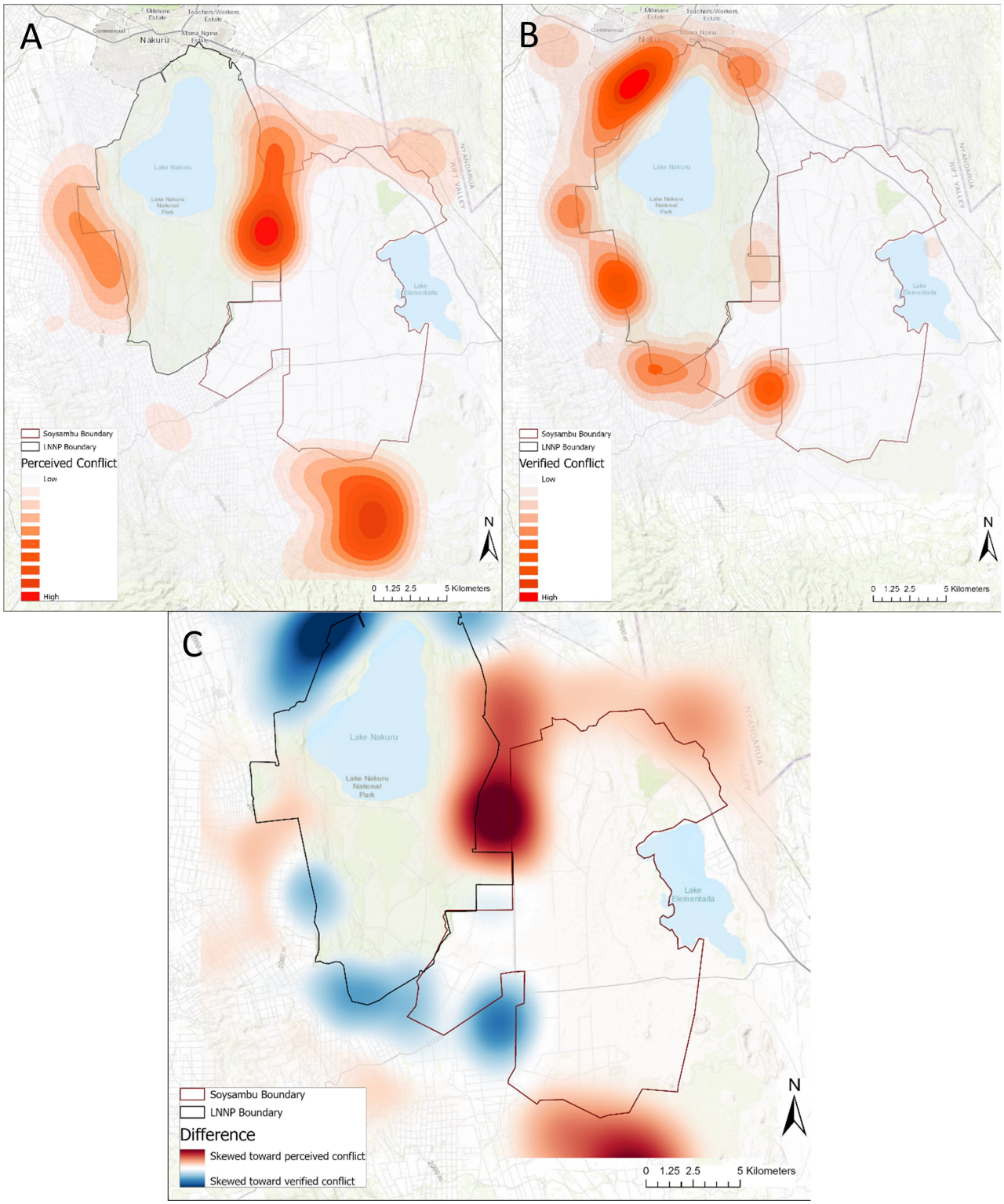


FIGURE 2 | Kernel density estimates of **(A)** perceived and **(B)** verified conflict reports, and **(C)** difference map showing $KDE_{perceived}$ - $KDE_{verfied}$.

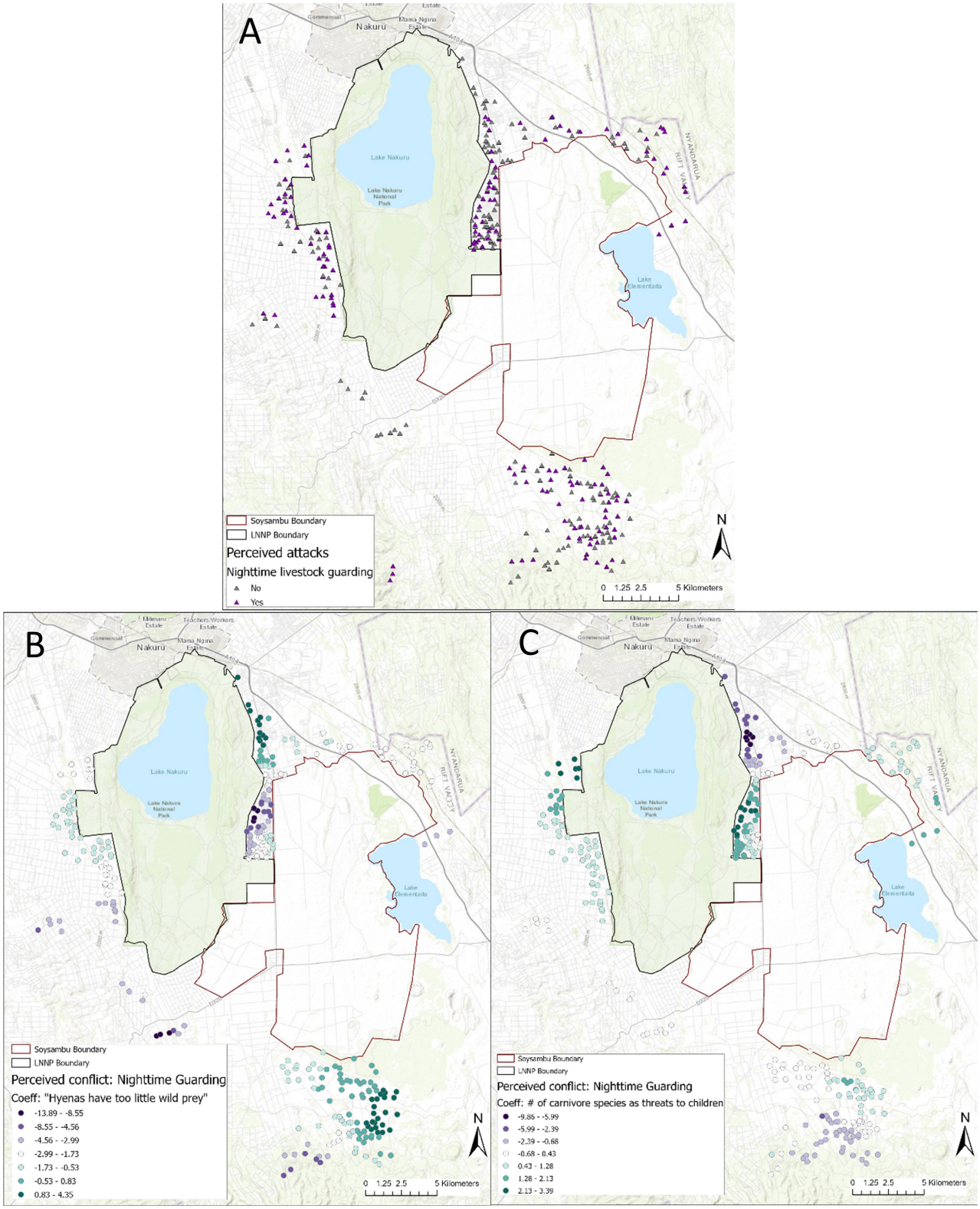


FIGURE 3 | (A) Perceived livestock reports color-coded by whether participant engaged in nighttime livestock guarding behavior, and geographically weighted logistic regression coefficients for clusters correlated with **(B)** belief that wild prey for hyenas is scarce, and **(C)** number of carnivore species perceived as threats to children, in relation to nighttime guarding behavior across locations of perceived livestock conflict.

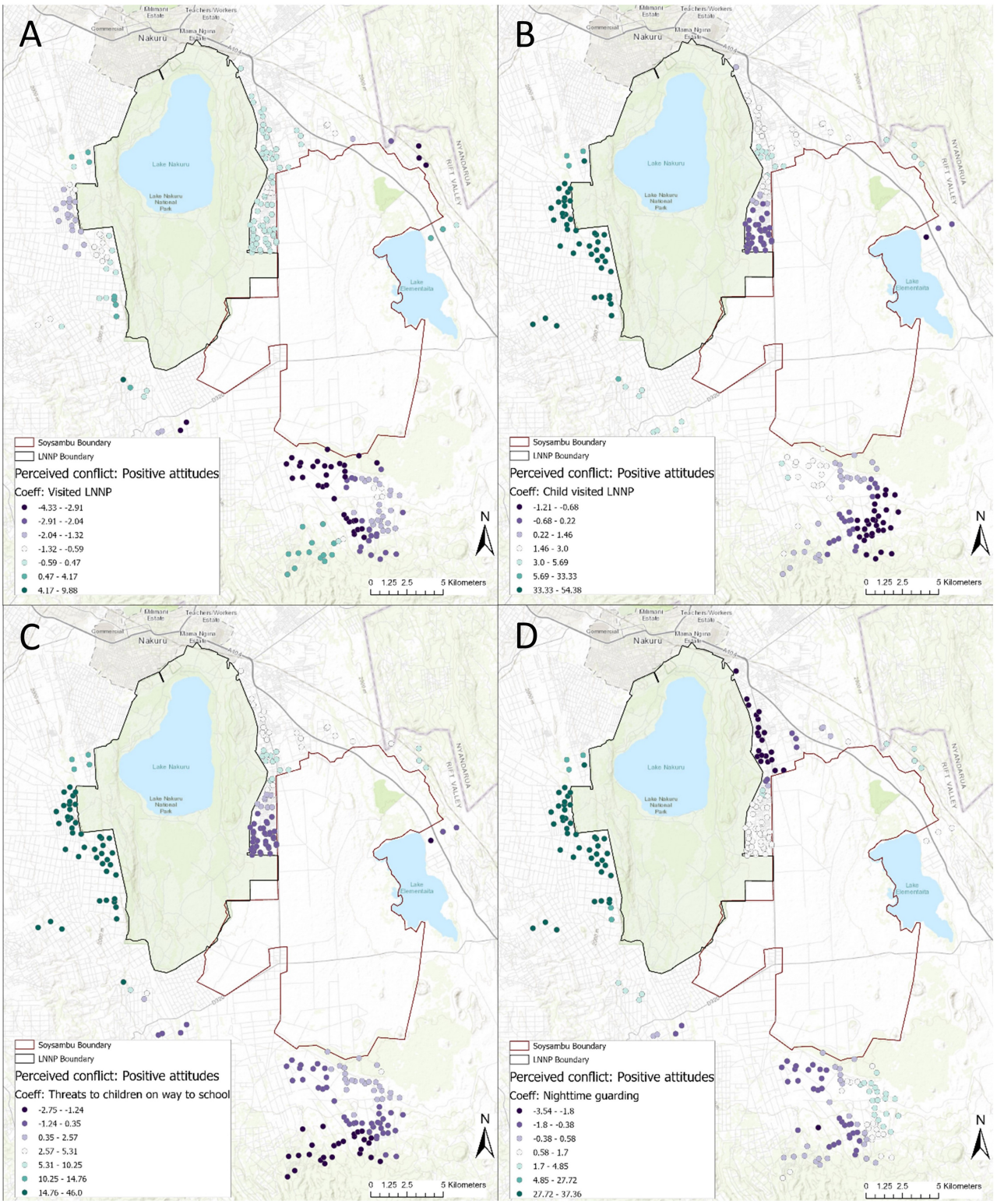


FIGURE 4 | Geographically weighted logistic regression coefficients for clusters correlated with **(A)** national park visitation by participant, **(B)** national park visitation by participant's child(ren), **(C)** whether participant perceives carnivore-related threats to their children on their way to school, and **(D)** nighttime livestock guarding, in relation to attitudes toward carnivores across locations of perceived livestock predation.

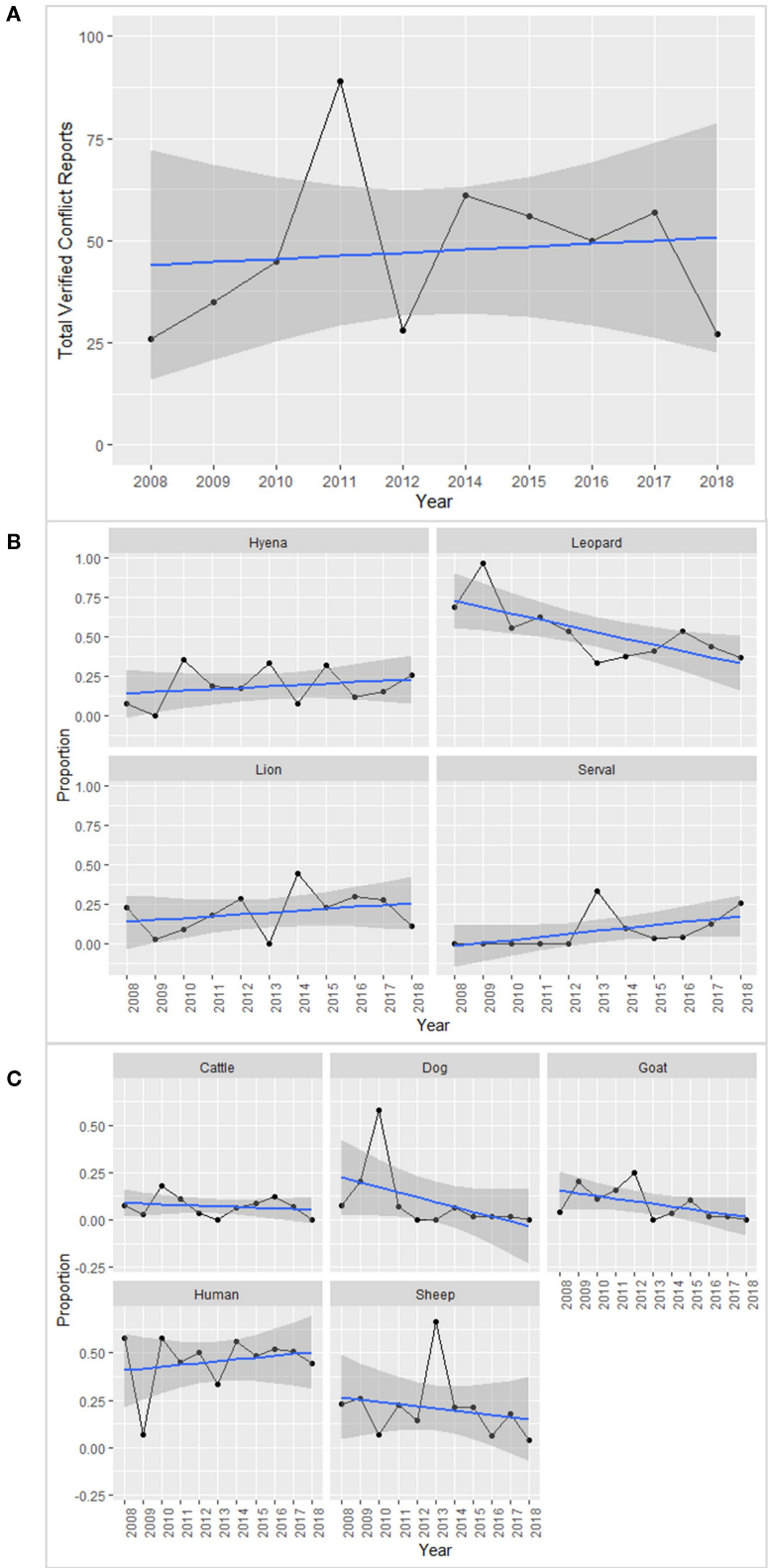


FIGURE 5 | (A) Total verified conflict reports, **(B)** relative proportions of carnivore species reported over time, and **(C)** relative proportions of reports regarding threats to livestock or humans over time for Nakuru County.

TABLE 3 | Statistically significant results of pairwise analyses examining predictors of perceived conflict reports.

	Perceived livestock attacks			
Variable	**Mean**	***p*-value**	***β***	**Test**
Gender				
Male	1.725	0.001		Wilcoxon signed-rank
Female	1.13			
Visited LNNP				
Yes	1.41	0.001		Wilcoxon signed-rank
No	0.82			
Education level		0.002	0.157	lm
Nighttime livestock guarding				
	1.289	0.064		One-way ANOVA
	1.484			

Predictors of Perceived Conflict/Risk and Attitudes Toward Carnivores

Gender and national park visitation were the strongest predictors of the number of conflict reports (**Table 3**). Pairwise analyses showed that livestock owners who have more children were less likely to indicate more perceived livestock attacks on the map ($\beta = -0.101$, $p < 0.05$). Livestock owners with a higher education level were slightly more likely to report more conflict events on the map ($\beta = 0.157$, $p < 0.01$). On average, men reported higher numbers ($\bar{x} = 1.725$) of perceived livestock attacks than women reported ($\bar{x} = 1.13$, $p = 0.001$). Unexpectedly, participants who had visited the national park were likely to report more perceived livestock attacks ($\bar{x} = 1.41$) than participants who had not visited the park ($\bar{x} = 0.82$, $p = 0.001$). If a participant guarded their livestock at night, they reported slightly fewer livestock attacks ($\bar{x} = 1.289$ livestock attacks) than those who did not actively guard their livestock at night ($\bar{x} = 1.484$), though the result showed low significance ($p = 0.064$).

When asked whether they feared risks to children from carnivores on their children's way to school, people who actively guard their livestock at night ($X^2 = 6.1274$, $p < 0.05$) or whose children walk to school [which is 86.2% of participants who had children] ($X^2 = 4.3355$, $p < 0.05$) were more likely to perceive risks to children. Carnivore species feared as risks to children were spotted hyena (34.4% of participants), leopard (33.9%), lion (27.5%), and black-backed jackal (20.1%).

Attitudes Toward Carnivore Conservation

Main Predictors of Attitudes

Education, national park visitation, and whether participants guarded their livestock at night were the strongest predictors of attitudes toward carnivore conservation (**Table 4**), with 70.8% of participants believing that spotted hyenas should be conserved. According to pairwise analyses on attitudes toward spotted hyena conservation (as a proxy for carnivore conservation more generally), if a participant had visited the national park, they were more likely to have positive views of hyena conservation than if they had not visited the park (Fisher test, two-sided, $p < 0.001$; **Figure 6A**). Additionally, if a participant's child had visited the park, they were more likely to have positive views of hyena conservation (Fisher test, two-sided, $p = 0.001$). This was true despite 82 participants in the latter group (i.e., 44% of the 186 participants with children who have visited the national park) never having visited the national park themselves. Attitudes toward hyena conservation were also more likely to be positive with increasing education level ($\beta = 0.3241$, $p < 0.001$; **Figure 6B**), and for livestock-owning participants who actively guard their livestock at night (Fisher test, two-sided, $p < 0.01$).

If a participant owned any species of livestock, they were less likely to believe hyenas should be conserved (Fisher test, two sided, $p = 0.01$). Participants who self-identified as farmers and herders for their primary livelihood (Fisher test, two sided, $p < 0.01$), or said their children face risks from carnivores on the way to school (Fisher test, two sided, $p = 0.01$), were also considerably less likely to report positive attitudes toward hyena conservation. Participants who owned at least one shoat in particular were significantly less likely to believe hyenas should be conserved ($\beta = -0.322$, $p < 0.01$), and cattle owners followed the same pattern ($\beta = -0.5011$, $p < 0.01$).

Reasons for Positive and Negative Attitudes Toward Hyena Conservation

Participants who said it was important for spotted hyenas to be conserved believed this due to ecotourism (83.3%), ecological reasons (43.2%), cultural reasons (15.2%), or other reasons such as for children to view in the future ("For the next generation"), or because hyenas were created by God ("They are God's creatures"). Participants who said it was not important for spotted hyenas to be conserved largely believed this due to the species' role in livestock attacks (89%), attacks on people (38%), or belief that hyenas are a bad omen (11%).

Of participants who supported hyena conservation, if the participant was older ($\beta = -0.2003$, $p < 0.01$), owned higher numbers of cattle ($\beta = -0.3439$, $p < 0.05$), or owned higher numbers of shoats ($\beta = -0.2372$, $p < 0.05$), they were less likely to say that ecotourism money was the reason to conserve spotted hyenas. Women ($X^2 = 4.1778$, df = 1, $p < 0.05$), participants with higher education levels ($\beta = 0.2279$, $p < 0.001$), and participants

TABLE 4 | Statistically significant results of pairwise analyses examining predictors of attitudes toward spotted hyena conservation.

	Hyena conservation		*p*-value	*β*	Statistical test
Visited LNNP	Yes	No			Fisher test
Yes	127 (48.1%)	34 (31.2%)	<0.001		
No	137 (51.9%)	75 (68.8%)			
Child visited LNNP					Fisher test
Yes	140 (53%)	45 (41.3%)	0.002		
No	103 (39%)	61 (56%)			
No children	21 (8%)	3 (2.7%)			
Livestock ownership					Fisher test
Yes	227 (86%)	98 (90%)	0.017		
No	37 (14%)	11 (10%)			
Shoat ownership					Fisher test
Yes	210 (79.5%)	90 (82.6%)	0.01		
No	54 (20.5%)	19 (17.4%)			
Education level			<0.0001	0.324	Logistic regression
Nighttime guarding					Fisher test
Yes	111 (42%)	44 (40.4%)	0.007		
No	115 (43.6%)	54 (49.5%)			
No livestock	38 (14.4%)	11 (10.1%)			
Fear risks from carnivores on child's way to school					Fisher test
Yes	145 (54.9%)	47 (43.1%)	0.011		
No	45 (17%)	26 (23.9%)			
No children in school	74 (28%)	36 (33%)			
Fear lions on child's way to school					Fisher test
Yes	83 (31.4%)	21 (19.3%)	0.034		
No	107 (40.5%)	52 (47.7%)			
No children in school	74 (28%)	36 (33%)			
Primary occupation: Farmer/herder					Fisher test
Yes	144 (54.5%)	77 (70.6%)	0.007		
No	120 (45.5%)	32 (29.4%)			
Livestock owners					
# of shoats owned			0.004	−0.501	Logistic regression
# of cattle owned			0.003	−0.322	Logistic regression
# of donkeys owned			0.012	−0.725	Logistic regression

whose children had visited the national park ($X^2 = 7.9898$, df = 2, $p < 0.05$) were more likely to report ecotourism money as a reason to conserve spotted hyenas.

Meanwhile, participants who had visited the national park were more likely to report ecological reasons to justify why it was important to conserve spotted hyenas ($X^2 = 4.637$, df = 1, $p < 0.05$), as were participants with higher education levels ($\beta = 0.1351$, $p < 0.05$).

DISCUSSION

This study used a uniquely interdisciplinary dataset to advance our understanding of the social and ecological drivers of human-wildlife conflict. Our analyses provided three main conclusions: (1) verified and perceived conflict exhibit quantifiably different spatial patterns, (2) information from verified conflict reports may be tied to anthropogenic ecosystem changes, and (3) park visitation, education level, and gender may be strong predictors of risk perceptions and attitudes toward carnivores, and can thus serve as conservation targets or mechanisms for managers in conjunction with spatial information.

Mismatch in Perceived and Verified Conflict

There were clear spatial differences between the perceived and verified conflict datasets. Areas of mismatch between verified and perceived conflict density comprised ∼20% of the conflict study area, with clear local regions where conflict skewed toward perceived or skewed toward verified. Though the effect was slim, NDVI was positively correlated with perceived conflict and negatively with verified conflict. This could be due to overinflation of perceived conflict in highly vegetated regions that

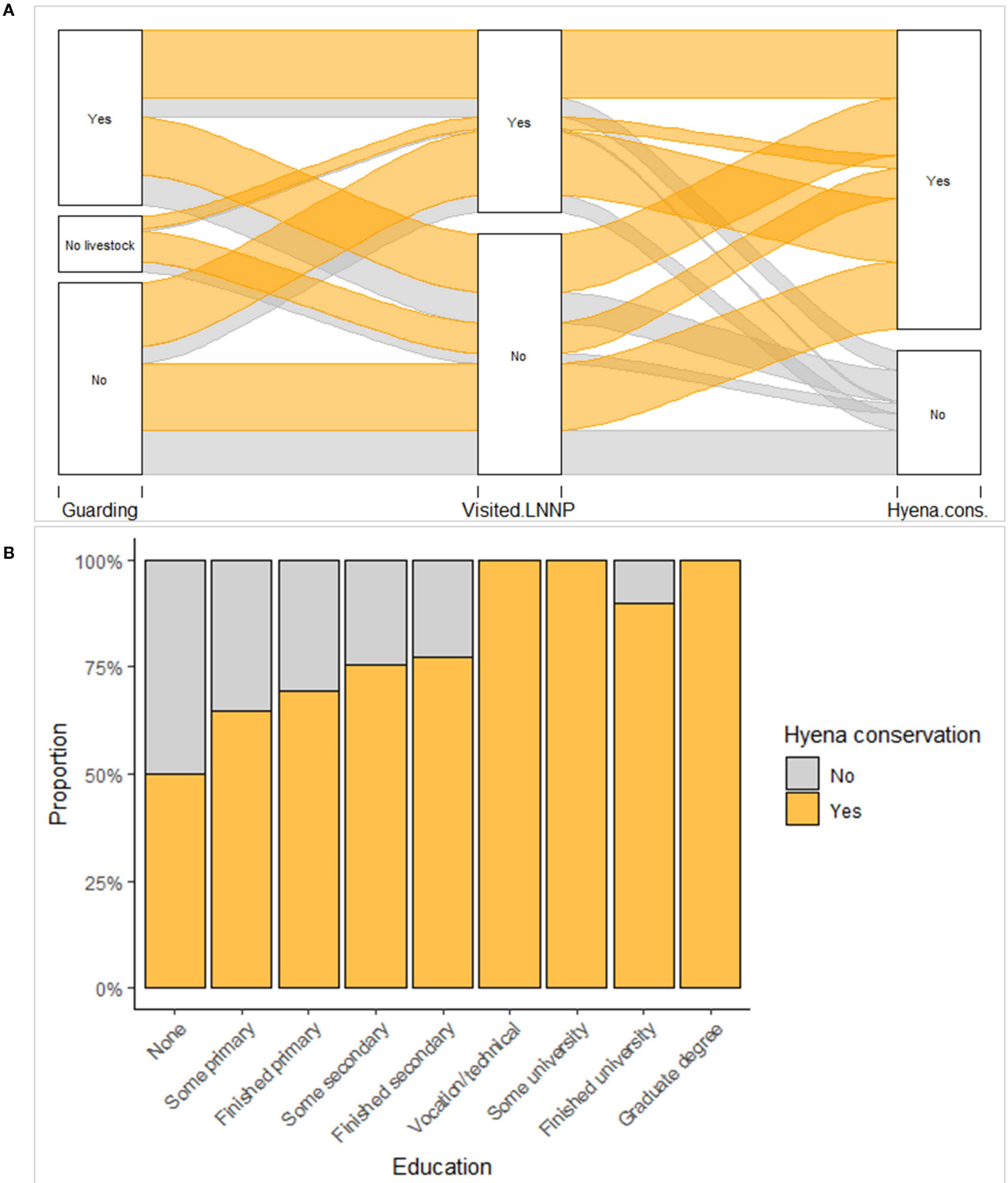


FIGURE 6 | (A) Frequency of participants answering the following questions: (1) Do you actively guard your livestock at night?, (2) Have you ever visited Lake Nakuru National Park, and (3) Do you think it is important to conserve the spotted hyena? Livestock guarding ($p < 0.01$) and park visitation ($p < 0.001$) were two of the most significant predictors of attitudes toward spotted hyenas. **(B)** Proportion of participants with varying education levels answering the question "do you think it is important to conserve the spotted hyena?".

carnivores could be more likely to use as habitat (e.g., Kolowksi and Holekamp, 2006; Thorn et al., 2012; Broekhuis et al., 2017).

Road density, meanwhile, was positively correlated with both verified and perceived conflict. This could be because where there are people, there are more roads, and in this region human population density is increasing due to a boom in immigration (Were et al., 2013). Because wildlife in this densely developed area are likely more nocturnal (Gaynor et al., 2018), people are likely experiencing carnivore conflicts at night near their homes (Ugarte et al., 2019), rather than during the day while animals are out to pasture in open or less road-dense areas. However, there is also broader literature showing that isolation of people from nocturnal animal activity may reduce conflict (e.g., Carter et al., 2012b), so further research is needed in this area. As we consider carnivore management in increasingly human-dominated landscapes, it is important to take into account how human activity and infrastructure is correlated with concentrations of conflict risk (e.g., Said et al., 2016), and translate these findings into thoughtful conservation-friendly infrastructure development.

While many people in our study reported using tools for nighttime livestock protection, livestock guarding was not a significant predictor of spatial patterns of perceived conflict. This aligns with our understanding that many HCC interventions are implemented without evidence of their effectiveness (Moreira-Arce et al., 2018). However, people's beliefs about two factors—-wild prey availability and carnivore-related threats to children—-correlated with spatial patterns of perceived conflict that was reported despite guarding efforts. Similarly, spatial trends of perceived conflict and their correlates were evident for regions where people still perceived livestock depredations despite their positive attitudes toward carnivores. These analyses can help us to understand not only what might compel people to over-report conflict, but also which regions to target for locally-specific drivers of conflict.

Trends in Verified Conflict

Verified human-carnivore conflict reports in Nakuru County exhibited several trends that ran counter to our predictions and may be a result of anthropogenic ecosystem change. For instance, seasonality was not a strong predictor of verified conflict, though the verified reports skewed slightly toward the dry season. This runs counter to a common belief that wildlife are able to disperse more widely during the rainy season in arid ecosystems (Koziarski et al., 2016), but correlates with reports of wildlife leaving fenced protected areas more frequently in the dry season (Kesch et al., 2015; Wilkinson et al., 2021).

Despite being one of the most abundant carnivore species in the region, black-backed jackals were not reported in the verified dataset. This result matched with the perceived data regarding participants' fears of carnivore-related threats to children on their way to school; jackals were the species least likely to be feared by participants. However, jackals are known to prey on vulnerable young livestock (Kamler et al., 2012), and have been seen doing so in this particular study area (author observation). Additionally, verified reports attributed to serval increased while leopard reports decreased. Because of the rapid development and deforestation in the region over the past decade (Mubea and Menz, 2012), it is possible that this trend is evidence of mesopredator release following declines in leopard populations (Prugh et al., 2009). While reporting bias and understaffing may have resulted in a limited verified conflict dataset from which to draw conclusions, the dataset's length of time and consistency of collection likely overcame these biases when assessing these basic trends. Future research on HCC should look more deeply into ecological and social drivers of observed trends in verified reports to better inform conflict management and to provide insight into broader ecological trends in conflict-prone regions.

Predictors of Perceived Conflict, Perceived Risk, and Attitudes

Visitation to Lake Nakuru National Park was one of the strongest predictors of lower perceived conflict, less perceived risk, and positive attitudes toward carnivores. This held true even if the participant themselves hadn't visited but their child had. Educational efforts regarding conservation are known for being frequently touted, but rarely evaluated (Tomicevic et al., 2010). Our results could be an important data point regarding the effectiveness of environmental education in communities dealing with conflict. This result is especially surprising given the intense immigration into the region; it is possible that visitation to the national park can drive formation of a "sense of place"—or connection to the environment in this region—and thus a stronger connection to wildlife. Sense of place (Hausmann et al., 2016) is solidified when people are young, which could be influencing the strength of the effect of children's national park visitation. Importantly, domestic tourism is not only important for connecting people with their protected areas, but is also one of many ways to address sub-Saharan Africa's over-reliance on international tourism which is subject to collapse during stochastic events such as COVID-19 (Lindsey et al., 2020). Kenya and other countries with similar reliance on tourism revenue could take these results as another benefit to enhancing their domestic tourism infrastructure to make protected areas more accessible to its citizens and particularly the local communities living near conservation areas (Sindiga, 1996; Okello et al., 2012).

While national park visitation could be influencing people's wildlife-related knowledge, formal education level was arguably the strongest predictor in our perceived conflict and attitude models. Our results show that having any amount of primary school education made a participant more likely to have positive views toward carnivores, and less likely to report perceived risk of livestock conflict. This aligns with other studies that have found education levels to be linked with positive attitudes and reduced risk perceptions regarding carnivores (Holmern et al., 2007; Dressel et al., 2014; Knopff et al., 2016; Koziarski et al., 2016). However, nearly 15% of our participants reported having received no schooling, which could be due to the lack of compulsory education during the schooling years of older participants (whereas now basic education in Kenya is compulsory and free), or in part due to school accessibility and transportation. In this region, some primary and secondary students are known to walk long distances to attend school (author observation). Education is an avenue for learning about the environment and perhaps changing attitudes toward wildlife (Tomicevic et al., 2010). The strong link we see between education and

perceptions of carnivores and conflict may be an additional compelling argument for increasing access to basic education and conservation awareness programs where communities are sharing landscapes with carnivores.

Lastly, across education levels and park visitation rates, gender played an important role in predicting the nature of perceptions and attitudes. For instance, women were more likely to voice that money from ecotourism was an important reason to conserve spotted hyenas. This could be because in many parts of sub-Saharan Africa, women serve as crucial links from the community to the national parks and reserves through selling handmade crafts and food to tourists (Twining-Ward et al., 2018). In fact, women participants in this study often enthusiastically reported they would "benefit from wildlife if we were able to sell our [goods] to tourists," even if they hadn't yet had the opportunity to do so. Notably, women also reported fewer perceived conflicts than men (i.e., less likely to over-report, and/or differences in daily experiences; Gore and Kahler, 2012), which could further reflect their importance as a specific demographic to target for co-created conflict solutions.

CONCLUSION

Human-carnivore conflict is a global challenge that is influenced by synergistic ecological and social dynamics. This study quantified differences in verified and perceived conflict and identified predictors of those differences. Despite the high levels of perceived conflict reported by interviewees, participants had largely positive attitudes toward carnivore conservation, even though there has been increased immigration into the region and a considerable subset of our interviewees were not long-term residents. Previous research has shown that the longer a person resides in the area, the more positively they feel toward certain species of large carnivores (Mkonyi et al., 2017), but our findings demonstrate more nuance in this than originally thought.

We were able to explore complexity in patterns of conflict using spatial analyses to understand where verified and conflict datasets diverge, what socioecological factors might predict spatial patterning in conflict reports, and which correlates of perceived conflict are more important in particular local regions. Our results provide empirical evidence to reinforce the understanding that working with communities to explore these mismatches can promote socially just and sustainable management of human-carnivore conflicts (Redpath et al., 2013). Additionally, our findings highlight the fact that land subdivision, fragmentation, and fencing within the landscape should be addressed through inclusive spatial planning to avoid exacerbating conflicts while supporting conservation measures and local community livelihoods (Said et al., 2016). Future research on human-carnivore conflict in developing landscapes should recognize that incorporating participatory methods and social science with ecological data is critical for inclusivity in addressing longstanding conservation conflicts and preventing the emergence of new ones (Weldemichel and Lein, 2019).

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved under UC Berkeley OPHS/CPHS Protocol #2018-02-10731. Verbal informed consent was retrieved according to the requirements of the institution and protocol. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

CW: conceptualization, methodology, formal analysis, investigation, writing—original draft, writing—review & editing, visualization, supervision, funding acquisition, and project administration. JB: conceptualization, writing—review, and editing. AB: methodology, project administration, writing—review, and editing. MK: conceptualization, validation, writing—review and editing, and visualization. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.681769/full#supplementary-material

Supplementary Figure 1 | Proportion of verified conflict reports seasonally over time. Dotted line demarcates 50% of reports.

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frontiers in Conservation Science

COMMUNITY CASE STUDY
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A Community Science Approach to Conservation Challenges Posed by Rebounding Marine Mammal Populations: Seal-Fishery Interactions in New England

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Between 1880–1962, gray and harbor seals were targeted in legal seal bounty hunts across Maine and Massachusetts due to a perceived competition with commercial fisheries. Following their extirpation 50 years ago, legislative protections allowed seals to recolonize historical grounds along the New England coast. With this conservation success story, conflict has re-emerged as seen in the numerous media articles reflecting a temperament beckoning to the past century, with calls to cull the population and the spread of misinformation. The return of seals after decades of near-absence has created a new ecological and psychological baseline for New Englanders where for three generations, seals were rarely present. Although seals are statutorily protected species, unlike the tools and resources available for depleted, threatened or endangered species, the support needed to increase opportunities for coexistence of humans with rebounding pinnipeds, are comparatively lacking. Even as gray seals have the highest fisheries bycatch levels of any marine mammal in the U.S., resources to address these management challenges are minimal due to limitations and prioritization processes for committing available support. While seal conservation has been a success, the manner in which management is often separately applied to ecosystem elements (e.g., harvested species, protected species) contributes to knowledge gaps, and a disconnect between the goals of conservation to sustainably utilize natural resources while also protecting the intrinsic value of resources for ecosystem health. Solutions to such coexistence challenges could benefit from a more holistic ecosystem conservation approach. To address these disconnects, a two-day workshop was convened to understand seal-fishery interactions where we provided opportunities for community members to meet and learn from one another including, but not limited to, fishermen, natural resource managers, marine mammal stranding response personnel and scientists. A convening that might otherwise result in tumultuous and adversarial engagement, we used as a tool to engage. This community science approach led to long-term relationships that have allowed for successful applied, and community driven, solutions.

Here we share the lessons learned and subsequent partnerships. Our intent is to share our approach to address other marine mammal conservation conflict challenges, allowing for collaborative pathways toward long-term coexistence.

Keywords: seals, fisheries, coexistence, community science, conflict, collaborative research

INTRODUCTION

Fishery interactions are a critical concern for marine mammal populations and ecosystem health, as well as a challenge for sustainable fisheries practices. Conflicts surrounding rebounding marine mammals and fisheries are common in marine systems (Nyhus, 2016, Guerra, 2019).

Interactions between marine mammals and fisheries are generally classified as operational (also referred to as direct or technical), in which marine mammals interact directly with a fishing operation; and indirect (or ecological), during which larger-scale competition or other trophic interactions occur among fisheries and marine mammal populations (Gulland, 1986; Plagányi and Butterworth, 2005). Depredation is a form of operational interaction in which marine mammals remove or damage fish captured in fishing gear, reducing the amount of landed or marketable catch. Depredation can have significant economic costs for some fisheries from lost/damaged catch and gear damage, and can cause injury or mortality to the predator from interaction with gear or deterrence methods (Read, 2005, 2008; Götz and Janik, 2013). Bycatch, or unintended catch of non-target species, is another form of operational interaction, currently the primary cause of serious injury and mortality for marine mammals globally (Read et al., 2006). Addressing issues surrounding bycatch and depredation will help maintain sustainable fisheries and practices as well as help address the challenges of recovering and rebounding pinniped populations. In order to best address the issue, actors from all sides of the conflict need to work together as collaborators to navigate and address interactions.

As ocean use increases and we consider truly embracing ecosystem-based management, addressing the challenges that are arising with rebounding populations of protected marine mammal species must become a priority (Roman et al., 2015, Cammen et al., 2019). If we do not, we may once again as in the era of seal bounty hunting, reach a sociological tipping point where politicized perception, rather than sound science and holistic ecosystem-based management objectives, will be sought to solve conflict. Dialogue must allow for full participation, sharing of knowledge and honest discussion that respects divergent views and perspectives to address the underlying conflict (Zimmermann et al., 2020).

As we attempt to address conflicts arising with rebounding populations of seals, community partnerships often need to have an opportunity created by trusted partners, where values can be respected. Community science applies the scientific method to social inquiry that is democratic, community-driven and community-controlled, characterized by place-based knowledge and social learning, collective action and empowerment (Salomon et al., 2018, Charles et al., 2020). This approach is necessary if we are to address conflicts and provide concrete and productive solutions for co-existence. We present here a case study of a community science approach taken to proactively address the seal-fishery conflict on Cape Cod, Massachusetts that is equitable, community-based and designed to respect all involved. The following are our experiences from convening this workshop, lessons learned, and progress made since our workshop, followed by recommendations for furthering these efforts.

Context in Which the Innovation Occurs

Nearly extirpated only 50 years ago, rebounding populations of gray and harbor seals are now a northeast U.S. marine mammal conservation success story. Between 1888 and 1962, an estimated 72,000 and 135,000 seals were reported killed by bounty hunters on the Maine and Massachusetts coast due to perceived competition with commercial fisheries (Lelli et al., 2009). As late as 1962, a five-dollar bounty was paid in Massachusetts per nose of each recovered seal. The bounty hunts devastated the U.S. populations of harbor and gray seals and resulted in the near extirpation of gray seals in the U.S. by the early 1960s (Katona et al., 1993; Wood et al., 2020). Following Massachusetts state protection 1965 and the passage of the Marine Mammal Protection Act (MMPA) in 1972, gray seals and harbor seals repopulated their historical grounds, with a current estimate of 27,000 gray seals and 75,000 harbor seals in U.S. waters (Hayes et al., 2019). As the first U.S. congressional legislation mandating an ecosystem-based approach to marine resource management, the primary objective of marine mammal management under the MMPA is to maintain the health and stability of the marine ecosystem. While conservationists celebrate this success, the increased presence of seals in New England has created unique challenges (Bogomolni et al., 2010; Roman et al., 2015; Jackman et al., 2018). For some, the populations of seals are "exploding," and the "seals are like vermin" (Boston Magazine, 2013; Nantucket Chronicle, 2013). Calls for culls and management action against seals have resumed, echoing similar sentiments of the 1800s.

The fishing community in New England has identified bycatch and depredation by harbor and gray seals as an issue of concern (Nichols et al., 2012; Rafferty et al., 2012). Gray seals in particular are of interest as their recolonization sites coincide with historically important fishing grounds in the region. According to the Northeast Fisheries Science Center (2020), "Fisheries interactions have also increased over the past two decades, with fewer than 10 total estimated gray seal interactions in 1993, to more than 1,000 annually in four out of the last five years; this is the highest bycatch of any U.S. marine mammal species."

In seal-fisheries conflicts, it is often the case that the limitation to finding solutions are impeded by adverse human-human

relationships and underlying presumptions directed at one group (of humans) over another (Pooley et al., 2017). While many of these challenges may ultimately be addressed by forming research questions and creating platforms to collect data to direct science, management and policy, the first step in addressing this conflict is to define who is in our community, and engage all actors from the beginning of these efforts (Redpath et al., 2013). Stakeholders in a process are actors (persons or organizations) with a vested interest (Schmeer, 1999). In this controversy, we look to the relationships between actors in our community including, but not limited to, fishermen[1], natural resource (marine mammal and fisheries) managers, marine mammal stranding response personnel and research scientists.

Detail to Understand Key Programmatic Elements

Community science partnerships allow the opportunity to provide technical support and knowledge collectively. Our approach to address seal bycatch and depredation was based on authentic engagement and community science over nearly a decade. As we developed opportunities for engagement over the years and the agenda for the 2015 workshop, we kept in mind four objectives: 1) Address the disconnect 2) Build trust 3) Build community, and most important 4) Commit to the long term.

1) **Address the Disconnect.** Community science requires an engagement approach with an emphasis on asking questions, listening to concerns, and doing so in an environment in which community members feel comfortable speaking freely.
2) **Build Trust.** We are all human and must acknowledge that each actor's professional persona belongs to an individual citizen with personal motivations, ideologies and experiences outside of ones' professional occupation. Meetings began with a clear outline of expectations for decorum, ample refreshments, and time for casual conversation.
3) **Build Community** It is important to bring science and management into community and community into the science and management. Our goal was not to pre-define the results or the recommendations that emerged. Listen to community discussions and from there, seek to fill knowledge gaps through building bridges between those with expertise and those with questions. All participants were valued and respected for their expertise and knowledge in their own fields. We designed a day-long series of presentations and hands-on activities that would facilitate learning, sharing and building relationships.
4) **Commit to the Long Term.** Recognize that all of the above three approaches require time, dedication, and investment in each relationship. Developing avenues for conversation shows commitment and desire to continue relationship building. This applies to the sciences as well.

[1]In the case study reported herein, fishing community members self-identified as 'fishermen' irrespective of their gender identity, so we have chosen to use their preferred term.

Engagement in these efforts began years before this workshop through a series of small meetings (Bogomolni et al., 2010). Following recommendations from these meetings, outreach was conducted in Cape Cod (Massachusetts, USA) fishing communities to connect researchers with commercial and recreational fishermen, building the foundation for cooperative research partnerships. Local researchers met with fishermen at harbors and community centers, volunteered as crew on fishing vessels, and attended relevant fishery association and management meetings. This collective and individual outreach effort was highly successful, laying the groundwork for numerous collaborative research projects involving fishing community partners. All research projects involved fishing community partners at the earliest stages of project development, which is key to building mutual trust and understanding among all parties, as well as laying the foundation for sound, hypothesis-driven science (Nichols, 2011).

Subsequently, a diverse group of over 60 commercial and recreational fishermen, scientists, and resource managers gathered at the Center for Coastal Studies (CCS) for the workshop, "Gulf of Maine Seals: Fisheries Interactions and Integrated Research," to discuss operational and ecological interactions between seals and fisheries. In order to ensure that the fishing community had a distinct voice, a forum was included in the workshop agenda, during which fishermen were encouraged to share their observations, experiences and concerns. The recommendations from all participants shared common themes, including the need for collaborative research involving both the scientific and fishing communities. Workshop organizers proposed a consortium to foster collaboration among managers, researchers and the fishing community (Nichols et al., 2012), which became the Northwest Atlantic Seal Research Consortium (nasrc.whoi.edu/sealconsortium.org). Our ongoing partnerships between scientists and fishermen studying seal/fisheries interactions (e.g., Nichols et al., 2014) helped leverage broader communication and collaboration across the fishing community, laying not only the foundation for future research, but also broadening the community conversations around issues of depredation, bycatch, and other interactions (Cammen et al., 2019). We worked to connect students with fishermen to conduct collaborative research projects on various aspects of seal/fisheries interactions, which added new perspectives to these community conversations (e.g., Sirak, 2015):

> "As a guest on the boat and a scientist, it is important to make compromises. You are already imposing on the fishermen by taking up extra room on the boat and by interfering with the standard fishing process at times. It is important to know when to draw the line. For example, I was able to stay out of the way while still counting each skate and bycaught animal that came up in the net. I was also able to record and photograph most of the damage done to the catch during the haul while still staying out of the way. However, I was not able to bring back any bycaught seals for necropsy as I originally had wanted to. While I asked the captain if we could bring bycaught seals back multiple times, I eventually realized that this was too much to ask of him. From the

BOX 1 | **Objectives of the Workshop**.

The primary objectives of the workshop were to:

1. Build a bridge between the fisheries, science and regulatory communities based on each group's expertise and experiences.
2. Gain a better understanding of the types of fisheries interactions that occur in the Gulf of Maine and Mid-Atlantic regions and how they may present (be visible) in case studies.
3. Discuss the benefits gained by all from providing marine mammal fishery bycatch to researchers, including the ability to learn about the overall health of a population, the animals' diets, and ecosystem health.

Through fulfilling these objectives, we strove to:

1. Reduce the adversarial nature between groups by working together to mitigate harmful marine animal/fishing gear interactions and learn from them when they do occur.
2. Work with regulators (through permitting process or other means) to increase positive relationships (e.g. collaborative research) that provide benefit to all groups.
3. Identify ways to reduce bycatch and injury to marine animals by the fishing community.

scientific perspective, each seal we dumped back overboard was an animal full of information we would never be able to get back, but from a fishermen's perspective each seal was a smelly waste of time that already damaged their gear and their catch and would take up too much space on the boat and draw too much attention from the tourists back in Chatham."—Laura Sirak, University of New England graduate student conducting research on seal bycatch and depredation (Sirak, 2015).

After extensive outreach and based on community feedback, we convened this two-day workshop to address disconnect and knowledge gaps among those involved in the fishing, marine mammal research and fisheries management (**Box 1**). The case study presented is based on our observations and insights. We intended to create a group learning experience in order to come to a greater common understanding of the causes and effects of, and lessons learned from, interactions between seals and fisheries, with a particular focus on knowledge gained from carcasses of bycatch.

The workshop took place December 1–2, 2015 in Chatham and Woods Hole, MA. The workshop locations were chosen with the intent of welcoming participants onto the "home turf" of the fishing and scientific communities, creating new experiences while also maintaining the comfort associated with surroundings familiar to community members. Individuals and/or organizations were invited based on previous engagement events or, as those identified as missing from previous discussions. This workshop consisted of a half day dedicated to introductory presentations to understand the perspectives and importance of marine mammals in the fishing, stranding and management communities, followed by a hands-on classroom and dockside practical to learn about fishing and gear from fishery observers and the fishing community, and concluded with an informal social gathering at a restaurant. The second day involved necropsies of stranded and bycaught seals with all attendees (including fishermen, scientists, marine mammal stranding responders and managers) and a synthesis of the workshop.

WORKSHOP APPROACH

Day 1 Agenda (Chatham Community Center, Chatham Fish Pier)

Following introductions, presentations were given on a range of topics, including assessment of injuries from fisheries interactions in stranded marine mammals, description of regional fisheries (e.g., gear types, fishing practices), rationale for and data collection in the fishery observer program, and fishing industry perspectives on seal/fishery interactions and collaborative research. Next, fishermen and fishery observer trainers led a hands-on demonstration and group discussion of fishing gear and fishing practices (**Figure 1**). Workshop attendees then traveled to the Chatham Fish Pier for a dockside examination and discussion onboard two commercial fishing vessels (**Figure 2**). Through in-person demonstrations, workshop attendees were able to develop a working understanding of fishing gear and fishing practices from experts in a manner otherwise unattainable in an academic setting.

Day 1 Summary

Despite the diverse and broad range of perspectives, participants expressed many common values, including the need for healthy marine ecosystems, concern for animal welfare, regulatory compliance, and the economic and cultural value of fisheries. The level of open, candid communication that took place during the afternoon and evening discussions indicated a great deal of mutual respect among participants and highlighted the value



FIGURE 1 | Workshop participants traveled to the Chatham Fish Pier for a dockside examination and discussion on board two commercial fishing vessels, a demersal longliner or "tub trawler" (F/V Noah) and a gillnetter (F/V Dawn T). Here, a commercial fisherman explains the inner workings of a gillnet fishing boat to several marine mammal researchers and marine mammal stranding network personnel.

of the workshop, designed to include dedicated opportunities for conversation, as a vehicle for building community and collaboration. Fishermen highlighted the importance of mutual respect as an incentive for increased communication with the fishing community, along with involvement of fishermen in research and management.

Fishing community participants suggested starting with a small group of researchers and fishermen to allow building of trust and mutual understanding. Among the specific ideas posed by the group was the potential application of novel acoustic deterrents to reduce interactions between seals and fishing gear, as well as specialized tools to safely release entangled seals from fishing gear while minimizing seal injury and gear damage. Some discussion centered on the availability of newly authorized acoustic deterrent technology and potential collaborative research to test its efficacy in local inshore fisheries.

Day 2 Agenda (Woods Hole Oceanographic Institution Marine Research Facility)

Participants followed safety protocols and were encouraged to actively participate in the necropsies of two bycaught seals and one stranded seal. This unique exercise enabled participants to: learn key morphological traits used to identify harbor and gray seals; assess carcasses to distinguish signs of fishery-related mortality from other causes of injury and illness; identify characteristics related to known gear interactions; share expertise and knowledge about what can be learned from obtaining whole carcasses for examination. The morning also provided a fun, open and inviting platform to get to know one another in a unique professional setting, which was followed by an afternoon discussion.



FIGURE 2 | Participants take part in the necropsy of stranded and bycaught seals. Here, a commercial fisherman works on opening the stomach of one of the bycaught gray seals recovered from a gillnet.

Day 2 Summary

The necropsy session included examination of seal stomach contents, leading to identification of prey (hake and squid) consumed by a harbor seal caught in a gillnet. Stranding personnel learned to identify fish prey with the help of fishermen and Northeast Fisheries Observer Program (NEFOP) staff and observers. There were several useful moments of unplanned information sharing, including an impromptu crash course for fishermen and observers on zoonotic pathogens and the simple strategies used by stranding personnel during field necropsies that might be useful for safely handling marine mammal bycatch aboard fishing vessels.

Following the necropsy, an afternoon discussion initially focused on operational interactions between seals and fisheries. Fishermen posed questions regarding seal diet, and whether seals are attracted to fishing activity or simply the food resources present around the fishing gear. Discussion from a management perspective included potential mitigation measures (i.e., acoustic deterrents) to reduce seal interactions. When asked about incentives for bringing seal carcasses in for study, or allowing researchers on board to study carcasses at sea, fishermen again highlighted the importance of starting with a small group to build mutual trust and understanding. The potential utility

of a specialized knife or other disentanglement tool to allow fishermen to safely free entangled seals without injury, in particular to avoid seals tearing out of gillnets while retaining a life-threatening "necklace" of netting, was discussed at length.

Fishermen noted that depredation of targeted catch was more costly than gear damage. The need was highlighted for more quantitative and qualitative, standardized data collection to document depredation and the associated catch loss. Studies quantifying depredation are extremely limited in the northeast (Rafferty et al., 2012; Sirak, 2015). Further discussion focused on studies of seal attraction and habituation to fishing gear, learning from bycatch events to inform modification to fishing gear, and the potential formation of a small, confidential working group of scientists and fishermen to study some of the above issues. It was pointed out that causes of seal/ fishery interactions likely involve multiple factors and consequently may require multiple mitigation measures. Fishermen expressed concern regarding the management implications of the findings of collaborative research, but embraced the value of building relationships between communities, sharing knowledge, and establishing trust. In order to inform public perceptions, the group agreed that more education was needed, and fishermen encouraged an interpretive presence at the Chatham Fish Pier.

Discussions between fishery observers and stranding network staff also proved fruitful. The group recognized that data collection on seals is not currently standardized among the two government programs, and discussions began regarding possible means to standardize data collection and analysis to increase the utility of both sets of data on seal injury and mortality. It was recognized that the fishery observer program's fishing gear training materials would be a useful resource for those assessing signs of injury in stranded carcasses, and that in general, cross-training between both programs would be extremely useful.

Methods to improve documentation of entangled/injured seals on haul-out sites and to better quantify injury and mortality using observer data were also discussed. Observer program staff expressed eagerness to collaborate and to provide data and samples, but also highlighted capacity limitations to collect data in field conditions, as well as the logistical challenges of bringing in samples, especially whole carcasses. Stranding network staff offered to assist with sample transport and storage. Representatives of both programs discussed sharing resources. In general, the need was highlighted for increased communication between all participants, and in particular among those in the government programs. Discussion also focused on human health issues and the challenges of messaging to the public and fishing community regarding risks and hazards associated with zoonotic disease.

Finally, the value in retrieving whole carcasses from fishery bycatch was highlighted. Studies of whole carcasses lead to improved understanding of seal diet, life history, and overall health. These cases could also be used to monitor ecosystem and animal health through surveillance of emerging pathogens and stressors of concern.

In order to benefit from what we learn from whole animal carcasses, involving fishermen and observers from the beginning of research was determined to be essential. Incentives for fishermen to bring back these animals to the dock with and without observers (under proper permitting) was discussed. Ideas from participants included more outreach, involvement, communication between all user groups. Ways to reduce negative public perception biases toward fishermen brought on by collaborative research of bycaught animals included simple solutions such as specially designed duffle bags to bring carcasses off vessels for research purposes.

DISCUSSION

Workshop participants stressed the need for cross-training and regular engagement as a way to share perspectives and experiences between communities. Some of the challenges described included understanding the pressures on fishing families and the larger industry, the need to address welfare considerations of bycaught marine mammals, and the regulatory guidelines which dictate the actions, and sometimes limitations, of management.

Specific recommendations included: standardization of data collection protocols and wider sharing of data related to anthropogenic injury or death to seals, increased opportunities to share knowledge across communities, retrieval of more whole carcasses from fishery bycatch, and working with marine mammal rehabilitation facilities to study live seals. Observations shared during group discussion included: the recognition that working with others to reduce seal bycatch and depredation benefits fishermen, and that fishermen want to be involved in research on and mitigation of seal-fisheries interactions. Many participants expressed positive feedback regarding this workshop, specifically on the objective of providing a successful forum to foster open communication and trust.

Collaborations Resulting From the Workshop

The value of this workshop can be seen in the diversity of participants engaging with each other years after the workshop. Many of these projects required time to obtain grants, permits and strengthen relationships. A letter of authorization was obtained from the NOAA NEFSC Protected Species Branch in Woods Hole for fishermen to recover seal carcasses resulting from bycatch events without having an observer on board. This initial step has had wide-reaching implications, as new data has been generated on seal diet relative to fishery catch, as well as the recovery of satellite-tagged seal carcasses along with the tags themselves. A team of fishermen and scientists began a project using multiple methods to document interactions between gillnets and marine life, including underwater video cameras mounted on fishing gear, on-deck documentation of depredation, and analysis of stomach contents from bycaught seals (Bogomolni et al., 2019). Similarly, other fishermen have begun to experiment with acoustic deterrent devices in commercial fisheries, in collaboration with researchers under guidelines from NOAA. Education and outreach recommendations from this meeting

included a new interpretive education collaboration at the Chatham Fish Pier between three organizations.

Acknowledgment of Any Conceptual or Methodological Constraints

Collaboration is a vital solution to coexistence. There are three factors that emerged which prevent more collaborative work to solve seal-fishery conflicts from taking place: 1) Power dynamics, 2) Risk Aversion and 3) Capacity.

Power Dynamics

A critical observation made during this process was that top-down approaches to conservation often create more conflict than seeking solutions inclusively from on-the-ground constituents (Lute et al., 2020). Organizational structure of conservation and resource management agencies is often compartmentalized, siloed and creates a power dynamic, hindering progress toward open dialogue and collaborative solutions to resolve conflict. Unless concerted effort is made for interdisciplinary collaboration and communication, conversations may be lacking critical perspective. In order to fully understand the challenges and develop solutions, improved communication and relationship-building between those who regulate activities pertaining to both seals and fisheries, as well as actors within their community, is needed. Such engagement amplifies the value of all actors and empowers the community to collaboratively participate in the problem-solving process. Rebounding marine mammal species create complex human dimension perception issues that require multidisciplinary and creative solutions at all levels of conflict (Pooley et al., 2017). Responsibilities of species protection and recovery should be coupled with amplifying the value and role of all community actors, empowering participants, and promoting innovative solutions that minimize conflict.

Risk Aversion

Perception of risk among and between stakeholder groups and individual actors can prevent action. Fears often prevent discussion or initiation of collaboration. For example, scientists may fear data being misinterpreted, stranding personnel may fear misunderstandings of their efforts toward animal welfare as dismissive to human welfare. Fishermen may sometimes be judged by other fishermen for working with scientists, conservation NGOs and managers. There may be a perception of risk if the public were to see fishermen bring in a dead seal to a public dock. There also may be a fear that science will be used against the fishermen in management. This mistrust in management can extend to mistrust in science, often conflating managers and scientists as the same entity.

Capacity

In the U.S., government funding directed toward particular marine mammal species is heavily weighted to those listed under the Endangered Species Act (ESA), or populations designated as depleted or categorized strategic under the MMPA, as well as assessing, monitoring, and minimizing threats to populations (Marine Mammal Commission, 2017), with little emphasis or support for interdisciplinary solutions needed to address rebounding protected species. While there have been decades to prepare for this predictable shift in baseline resulting from species protection and recovery efforts, less effort from management agencies has focused on developing mechanisms for the community to resolve conflicts that arise, increasing divisiveness (Guerra, 2019). Similarly, increasing the value of community science and authentic interdisciplinary partnerships would also increase capacity. This could translate as an increase in human dimension research support, in conjunction with applied scientific research to value expertise, knowledge, and a community science approach for rebounding protected species issues.

Supporting partners in this effort also means recognizing the need for general operating budgets, direct compensation and flexible timelines to address challenges and costs associated with vessel and equipment maintenance, weather delays, and unpredictable fishery dynamics (species abundance/distribution). Researchers need to account for the time it takes to engage with the community, and fishermen may require compensation for lost fishing opportunities.

Recommendations

In order to address rebounding marine mammal-fishery conservation conflicts, we recommend the following:

1) **Address the Disconnect.** The best lesson learned is one that serves us all well, "just listen." It is easy to be ready with an answer, a solution pre-baked before a conversation is even initiated. Listen, and be ready to be uncomfortable. Understand and acknowledge who is missing at the table as well as who is present, and strive for more inclusion. Ensure all actors have the opportunity to be heard and recognized. Scientists do not necessarily have facilitation training. Enlist a professional facilitator, work with social scientists and increase the skillset among scientists to include facilitation, communication, and conflict resolution.
2) **Create Opportunity.** Establish more opportunities for actors to engage with each other in a manner that is perceived as a shared benefit in knowledge. Create avenues to put new ideas into process, and opportunities to create dialogue to increase tolerance and mutual respect. Allow the opportunity for community partners to participate in scientific research, as well as for scientists to understand how local knowledge is gained. In our example, we recognized that decisions about fishing interactions with marine mammals were being made by some actors without an understanding of fishing practices. We also recognized that there was little opportunity for fishermen to interact directly with marine mammals and those who work with them in a way that was not about enforcement or provided beneficial knowledge.
3) **Communicate and Commit.** Initiate dialogue and follow through. It is the responsibility of workshop conveners to facilitate and clarify perspectives when appropriate. This open communication can help to dispel myths and improve information accuracy. As a convener, do not initiate recommendations without transparent communication among all participants throughout the process. As

recommendations become actions, this ensures that all actors' input has been considered.

4) **Increase Awareness**. Bring the challenges identified by community to the attention of the public, policymakers and government. Appropriation of funding to address rebounding marine mammal species conflict issues could help foster collaborations. These efforts can be defined within the goals of the MMPA to maintain ecological stability through an ecosystem based management framework. This should include a high priority on human dimension research, education, outreach and efforts to mitigate bycatch.

CONCLUSIONS

While on the surface the seal-fishery conflict described herein presents solely as a human-animal conflict issue, the challenge is heightened by the tensions within and between human communities (Nyhus, 2016; Guerra, 2019; Blount-Hill, 2021). Like most human-wildlife conflicts, the issue begins with biases and assumptions, perceptions, and in this case, the historical context of the dilemma (Dickman, 2010; Madden and McQuinn, 2015; Burt et al., 2020). When conservation success results in rapid change, some actors will seek familiarity, standing firm on a baseline founded upon what was known in their cumulative lifetime, and not current or emergent situations. Distrust then leads to decreasing engagement and hampering seeking of and implementing solutions to reduce conflicts (Cook, 2015).

To overcome distrust, marine conservation challenges require commitment. The groundwork for this workshop began years before implementation. A comprehensive approach to addressing seal issues started in 2006, nearly a decade prior to this meeting, with local fishing partners reaching out to seal scientists in response to concerns by commercial fishermen and an increase in seals. The workshop discussed herein took place in 2015, and we are currently planning a follow-up workshop (2021) to continue discussions, integrate new actors, update information and update strategies. This long term commitment was critical for past success and will continue to be for the future.

Conservation success is dependent on human behaviors and investment and coordination within local communities. Management of natural resources and their ecosystems require broad and meaningful community engagement, and decision-making pertaining to the conservation and protection of natural resources and ecosystems needs to be inclusive of the communities living, operating, and relying economically on resources within these ecosystems. Funding opportunities that incentivize inclusivity, both in the context of methods and results, of members of the local communities add value to, and elevate regional impact of, the work being conducted. Collaborative research programs and requirements for public outreach in planned activities provide opportunities for increased community engagement through communication and connecting the science, as well as increasing interest and awareness; however, conservation solutions for coexistence need involvement of the people within an ecosystem at all stages of planning and implementation process in order to be successful.

Coexistence is not the absence of conflict, and where there are humans with different experiences, and with different attitudes and perceptions about seals, there will be conflict. However, there are hopeful changes in how we address and perceive these challenges. During a recent webinar presentation on seals (Seacoast Science Center, 2020), the lead panelist from the marine mammal stranding field recounted their experience from this workshop six years prior where they watched a commercial fisherman open the stomach of a bycaught seal, betting to find a belly full of his target species (cod), only to discover a non-target-species (hake). While this fisherman lost the bet, in that moment, everyone won a new appreciation, and a concrete experience that continues to be shared with others, allowing the conversation to shift, and the visualization of a new way to work toward coexistence.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

ETHICS STATEMENT

Written informed consent was obtained from the individual(s) for the publication of any potentially identifiable images or data included in this article.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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Social Effectiveness and Human-Wildlife Conflict: Linking the Ecological Effectiveness and Social Acceptability of Livestock Protection Tools

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Human-wildlife interactions are embedded within socio-ecological systems (SES), in which animal behavior and human decision-making reciprocally interact. While a growing body of research addresses specific social and ecological elements of human-wildlife interactions, including conflicts, integrating these approaches is essential for identifying practical and effective solutions. Carnivore predation on livestock can threaten human livelihoods, weaken relationships among stakeholders, and precipitate carnivore declines. As carnivores have received greater protection in recent decades, researchers and managers have sought non-lethal tools to reduce predation and promote coexistence between livestock producers and carnivores. For these tools to be successful, they must effectively deter carnivores, and they must also be adopted by producers. Relatively few studies examine the practical and context-specific effectiveness of non-lethal tools, and even fewer simultaneously consider their social acceptability among producers. To address this gap, we suggest that a tool's ecological effectiveness and social acceptability be analyzed concurrently to determine its *social effectiveness*. We thus paired an experimental study of a carnivore predation deterrent called Foxlights® with qualitative interviews of livestock producers in Northern California. We placed camera traps in sheep pastures to measure the response of coyotes (*Canis latrans*) to experimentally deployed Foxlights and interviewed livestock producers before and after the experiment. Our experiment revealed weak evidence for reducing coyote activity with Foxlights, but interviews revealed that the potential adoption of tools had as much to do with their social acceptability and implementation feasibility as with evidence-based measurements of tool effectiveness. Interviewees viewed Foxlights as potentially effective components of husbandry systems, despite the data suggesting otherwise, demonstrating that scientific reductionism may lag behind producer practices of systems-thinking and that isolated demonstrations of a tool's ecological effectiveness

do not drive tool adoption. Future empirical tests of non-lethal tools should better consider producers' perspectives and acknowledge that data-based tests of ecological effectiveness alone have a limited place in producer decision-making. Iteratively working with producers can build trust in scientific outputs through the research process itself.

Keywords: human-wildlife conflict, human-wildlife interactions, conservation planning, monitoring and evaluation, human dimensions of wildlife, conservation social science, non-lethal control, socio-ecological system

INTRODUCTION

Human-wildlife conflict (HWC) can drive wildlife declines and threaten human livelihoods. Large carnivores are particularly susceptible to declines due to conflict because their large ranges, carnivorous diets, and adaptability have put them into frequent contact with people (Ripple et al., 2014; Wolf and Ripple, 2017). The loss of these species can in turn transform ecosystems and trigger collapses (Estes et al., 2011). HWC and the coupled human-natural systems in which conflicts occur are driven by a dynamic array of interconnected social and ecological elements, in what is referred to as socio-ecological systems or SES (Berkes and Folke, 2003; Ostrom, 2009; Lischka et al., 2018). For example, while the behavioral and spatial dynamics of carnivores and their livestock prey may be understood through an ecological lens (Wilkinson et al., 2020), the arena in which these species encounter one another is shaped by past and current land and livestock management practices that are selected through separate and complex social, political and economic processes. Conflict poses considerable challenges for those who bear the costs associated with carnivore conservation (Muhly and Musiani, 2009) and is deeply embedded within the value systems and identities of people who have personal and family histories in agricultural production (Widman and Elofsson, 2018). The traditional roles that conflict management has played in agricultural contexts have been profoundly meaningful, and the symbolic threat of carnivores can be as important as economic hardship in dictating the terms of conflict (Skogen et al., 2019). Thus, integrating the disparate elements of HWC and the feedbacks that link them requires transcending the barriers that have traditionally divided social and bio-physical sciences (Dickman, 2010; Redpath et al., 2012).

There has been a push for applied research on tools to mitigate conflict, but much of this research misses the socio-ecological nature of the problem. For example, livestock-carnivore conflict is one of the most pervasive forms of HWC (Inskip and Zimmermann, 2009; Lute et al., 2018) and management strategies in North America have long relied on lethal strategies aiming to reduce carnivore numbers or eradicate them completely (Reynolds and Tapper, 1996; Berger, 2006; Barnes, 2015). These strategies have recently become less viable for a variety of social reasons (Berger, 2006; McManus et al., 2015; Slagle et al., 2017; Lute et al., 2018) and ecological reasons (Bergstrom, 2017; Lennox et al., 2018; Moreira-Arce et al., 2018). As many carnivore populations in the United States are recovering, wildlife managers and livestock producers require new strategies to protect both livestock and carnivores.

Non-lethal livestock protection has become a central focus of a growing body of research dedicated to carnivore conservation. Research suggests that non-lethal strategies may protect livestock as well as or better than lethal strategies, and there has been an effort to understand their effectiveness by ecological metrics (Miller et al., 2016; Treves et al., 2016; Eklund et al., 2017; Moreira-Arce et al., 2018; van Eeden et al., 2018). Perhaps more importantly, many livestock producers believe that non-lethal strategies are only slightly effective at best and seldom long-lasting (Scasta et al., 2017), which has prompted researchers to call for new empirical studies to convince stakeholders of the value of non-lethal approaches. But these calls often assume that the adoption of tools by stakeholders is singularly guided by their access to conclusive science.

Social acceptability is an important dimension of these non-lethal conflict mitigation tools, as the effectiveness of a tool matters little if producers do not use it. While it is possible that empirical demonstrations of effectiveness may lead to greater adoption of non-lethal strategies (Baker et al., 2008), producers' decisions are not usually informed by academic research (Knapp and Fernandez-Gimenez, 2009) and scientific evidence is often contested or dismissed when social conflict is intense (Woodroffe and Redpath, 2015). Despite the growing scientific understanding of the ecological effectiveness of livestock protection tools, it is unclear whether and how this expanding body of literature influences which tools producers use, as the limits of a tool's applications are also driven by attitudes, values, context, and social networks (Wilmer and Fernández-Giménez, 2015; Pooley et al., 2016; Lozano et al., 2019). Broadening the definition of effectiveness to necessarily include the willingness of stakeholders to adopt tools will require a better understanding of how and why livestock producers make husbandry decisions, how knowledge is transferred and evaluated, and what social and ecological elements inform the social acceptability of a tool. Areas where attitudes and scientific data diverge indicate targets for stakeholder engagement and collaboration.

To investigate how the ecological effectiveness and social acceptability of a non-lethal tool interact to inform an integrated metric of *social effectiveness*, we paired an experiment testing the ecological effectiveness of a predation deterrent (Foxlights®; Bexley North, Australia) with livestock producer interviews in Northern California. We conducted qualitative interviews both before and after sharing the scientific results of the non-lethal tool's ecological effectiveness because this can be a powerful way to examine how science is integrated into a producer's decision-making process (Drury et al., 2011; Wutich et al., 2019; Martin, 2020). Foxlights are predation deterrents that flash randomly

timed and colored lights in all directions from sundown to sunup to mimic lights that are associated with human presence, and are designed to be used based on line-of-sight. We chose Foxlights because of their reported ecological effectiveness (Ohrens et al., 2019a; Naha et al., 2020) and growing popularity. We evaluated the effects of Foxlights on coyote (*Canis latrans*) activity in a sheep production operation in Northern California, as coyotes pose the most significant predation risk in this geographical context (USDA, 2015).

We then sought to reciprocally combine our ecological examination of Foxlights with our qualitative approach to estimating social acceptability to produce an integrated socio-ecological understanding of tool adoption. In addition to testing whether Foxlights reduced local coyote activity, we aimed to better understand how producers make decisions, the role empirical science plays in that process, and what other socio-ecological factors serve as opportunities and barriers to tool adoption. We also examined whether our iterative integration of stakeholder knowledge improved receptivity to empirical findings and improved the trustworthiness of both the research and researchers. This situation assessment serves multiple goals, as it can be used to inform the monitoring and evaluation component of a planning cycle, provide a new and transdisciplinary approach to tool evaluation, and reveal how stakeholders may respond to tool recommendations. In the following sections, we will draw from various theories in the field to explain how ecological effectiveness and social acceptability can be used to define *social effectiveness*, describe our qualitative and empirical methods, present the results of the interviews and the Foxlights study, and then summarize how the empirical study and the interviewees' perspectives demonstrate the value of a systems-oriented approach to tool evaluation that accounts for social effectiveness.

THEORETICAL FOUNDATIONS OF SOCIAL EFFECTIVENESS

Multiple theoretical perspectives guided our research. We primarily used a socio-ecological systems (SES) approach to evaluate a given tool's ability to mitigate conflict and promote coexistence, acknowledging that human and animal behaviors are informed by both social and ecological dynamics and feedbacks. Thus, we viewed producer-carnivore conflict as an interaction of humans and animals, whose respective attributes and behaviors have co-developed across overlapping spatial and temporal scales. We used the definition presented by Carter and Linnell (2016) to understand coexistence as a "dynamic but sustainable state" that involves adjusting human interactions with wildlife to ensure co-adaptation, suggesting that coexistence with wildlife requires more intention than merely existing in the same place at the same time. Just as Lischka et al. (2018) accounted for the bidirectional impacts of social and ecological processes on black bear conflict with homeowners, we too acknowledged the individual agency of both producers and coyotes as well as wide-ranging external influences on their behavior. For example, coyote presence in Northern California is impacted by ecosystem characteristics, such as topography and prey abundance, as well as by societal drivers that include tolerance for coyotes (Bruskotter and Wilson, 2014) and patterns of human development. This SES approach thus highlights the need to understand both the social and ecological factors that contribute to conflict and, most importantly, conflict-mitigation.

We propose the term *social effectiveness* that incorporates both ecological effectiveness and social acceptability. An examination of a tool's social effectiveness will fill multiple lacunae in the field of HWC. Currently, not enough is known about which tools are ecologically effective, even less is known about tools' social acceptability, and the field is lacking work that addresses both of these questions simultaneously (Hartel et al., 2019). In our study, we defined ecological effectiveness as the ability of Foxlights to deter coyotes from pastures. We defined social acceptability following Shindler and Brunson (2004) as an ever-evolving process that helps determine the adoption of any particular policy, program, or tool. Social acceptability is not an active area of research within HWC, but several theories suggest its potential importance to this field and point to the need for empirical research on the topic. These theories include hazard acceptance models, human dimensions of wildlife, taskscapes, and diffusion theory, among others. Key components of social acceptability identified both in our research and others include social trust, values and attitudes, context and systems, information transfer, and the research process itself. Here we define these key components as they relate to our study.

Social Trust

Social trust is a major, if not the major, component of social acceptability. Given that HWC does not always involve human conflicts with wildlife but can also entail conflicts between humans over wildlife conservation issues (Redpath et al., 2015; Slagle and Bruskotter, 2019), it follows that social trust among stakeholders can override all factors when it comes to determining the social acceptability of a proposed solution (Shindler and Brunson, 2004). Social trust is defined as a decision-making heuristic that involves conferring some responsibility to an outside entity for things out of one's control, and can be used to examine perceptions of risk and acceptance of new technology (Siegrist, 2000; Siegrist et al., 2000). It is an adaptive process that takes time, requires multiple opportunities for interaction, and is linked with knowledge, honesty, and care (Peters et al., 1997). Social trust is a primary component of the hazard-acceptance model, a psychological model that claims that tolerance for large carnivores is informed by an array of factors including social trust, affect for species, risk perceptions, and tradeoffs (Bruskotter and Wilson, 2014; Slagle and Bruskotter, 2019). In their definition of social acceptability, Shindler and Brunson (2004) identify many of the elements of the hazard acceptance model without naming the term. Given the role of social trust in determining individuals' willingness to rely on external decision-makers, it follows that trust for the researchers studying a particular tool could lead to lower perceived risk in adopting the tool. To build social trust, Bruskotter and Wilson (2014) recommend highlighting shared fundamental values and goals, and Shindler and Brunson (2004)

emphasize the importance of cumulative interactions over time. Social trust is especially pertinent to our study, as the social acceptability of a tool is ultimately individually determined (though informed by broader cultural and ecological contexts), and we forged trust with the individual producers over the course of iterative interviews.

Attitudes and Values

Social acceptability is also conditioned by attitudes and values. Here we draw from theories in the field of human dimensions of wildlife (HDW). In particular, we build on research that considers the roles of attitudes and values in informing why humans behave the way they do with regard to wildlife, what human behaviors lead to conflicts, and how human behavior might be influenced to minimize conflict (Manfredo et al., 1995; Decker et al., 2012; Dietsch et al., 2019; Hiroyasu et al., 2019). Attitudes are favorable or unfavorable dispositions toward an action. For example, a positive attitude toward carnivores may explain behaviors like reluctance to employ lethal means of carnivore control. Attitudes are in turn guided by values, which are fundamental, consistent belief systems that transcend specific situations. For example, one's positive attitude toward carnivores may be based on values of mutualism, which is a belief system associated with egalitarian views of wildlife and a conviction that human activity should be limited for the sake of wildlife protection (Manfredo et al., 2017). Attitudes and values are unique to individuals and inform their identities, but values also exist along a continuum and can reflect broader shifts among groups of people. An example is the Western post-WWII movement toward mutualist values from more traditional domination values, which are linked to a belief that wildlife exist for human use. This value shift has resulted in a recent backlash among those with traditional wildlife values, often out of a desire to protect cultural heritage. Manfredo et al. (2017) revealed through a 19-state survey that in states like California that tend toward mutualist values, the potential for social conflict over wildlife issues with people who have domination values was much higher. It follows that carnivores can become emblematic of greater change as their presence becomes further mired within contested values.

In considering social acceptability, we therefore also draw from theories that describe the symbolic roles carnivores play in determining attitudes and values. In particular, our research builds on the theory of "taskscapes," which involves looking at how a landscape is understood by the histories and identities connected to the work and play people undertake in it (Ingold, 2000; Skogen et al., 2019). People are generally more concerned by taskscape changes, or changes to how a landscape is used, than by physical landscape changes. Carnivores can become symbols of greater taskscape change if the changes that bring carnivores are perceived as being imposed by threatening external forces, meaning that anti-carnivore attitudes can develop independently of material costs. On the other hand, positive attitudes toward the changes that bring carnivores may foster tolerance as long as material damage is not extensive. Approaches like non-lethal tools aim to concurrently help producers achieve livelihood goals *and* promote carnivore conservation. Thus, the attitudes producers hold toward these tools may be linked to their attitudes toward carnivores and all that carnivores symbolize within a taskscape.

Context and Systems

Social acceptability can be specific to a given context, as a solution or tool that is appropriate in one system may not be appropriate in another (Shindler and Brunson, 2004). For example, the heterogeneity of ranch characteristics and ecoregions in combination with individual producer attributes may mean that no single solution can satisfy the diverse needs of varied ranch operations (Roche et al., 2015). A systems approach can thus help account for the complexity of a producer's decision-making process by acknowledging that social acceptability does not exist in a vacuum; it is instead in relation to what the alternative solutions are perceived to be within a given context. For example, producers operating on privately-leased land often have a different set of alternatives than those on publicly-leased land. Brunson (1996) defined social acceptability as a "condition that results from a judgmental process by which individuals (1) compare the perceived reality with its known alternatives, and (2) decide whether the real condition is superior, or sufficiently similar, to the most favorable alternative condition." Alternatives are difficult to articulate simply, although this is often what is done when alternatives are presented to producers. For example, non-lethal methods and lethal methods are often presented as alternatives to each other, even though they can be employed simultaneously. This renders social acceptability a dynamic and potentially "wicked" process that changes with available alternatives (Rittel and Webber, 1973; Whyte and Thompson, 2012; Mertens, 2015). What may have been acceptable in the past can become unacceptable in the future. This dynamism further underscores the need for an iterative process of stakeholder outreach in order to continue to assess social acceptability as both context and systems evolve.

Information Transfer and Research Process

The way that information is transferred as well as the research process itself also informs social acceptability. Studies have found that producers primarily get their information via word-of-mouth, especially from neighbors and other producers, as opposed to technical sources (Rowan et al., 1994; Kachergis et al., 2013; Roche et al., 2015). For example, diffusion theory presents producers as rational actors who utilize social networks to make decisions. This theory supports the finding that the adoption of new technologies generally begins with opinion leaders, who are producers that are well-connected within knowledge networks (Lubell et al., 2013). These opinion leaders then pass on new technologies to others in their networks, or new technologies are passed down through ranch family generations. A tool's social acceptability can thus be influenced by the way a producer learns about the tool and who they learn this information from. Furthermore, the way in which people are incorporated into a decision-making process can influence their attitudes and judgements (Shindler and Brunson, 2004). Thus, transdisciplinary approaches that emphasize producer involvement may contribute to the social acceptability of research

findings, and such approaches have been called for by previous researchers (Hartel et al., 2019). Disciplinary or interdisciplinary approaches on their own are not always flexible enough to be able to address real-world problems because they do not incorporate non-academic actors. Conversely, transdisciplinary approaches aim to identify solutions via a process of cocreation by incorporating differing values and perspectives that better reflect the de facto decision-making process. These collaborative efforts enable researchers to span multiple social networks and coproduce knowledge with livestock producers who can contribute their own diverse epistemic backgrounds.

Ecological Effectiveness

We based our metrics of ecological effectiveness on several reviews that systemically evaluated experiments on lethal and nonlethal livestock protection methods (Miller et al., 2016; Treves et al., 2016; Eklund et al., 2017; van Eeden et al., 2018). These reviews sought to determine which interventions work best, and generally defined ecological effectiveness as the change in livestock loss or carnivore presence in pastures before and after techniques were applied or between control and treatment groups. None of the reviews were able to make any definitive claims and determine "what works" due to a lack of robust studies. Given that the field of HWC lacks a consistent standard for evidence of ecological effectiveness, the authors of these reviews have called for future examinations of ecological effectiveness to satisfy a "gold standard" of scientific rigor that pays special attention to controls, randomization, and replication.

To achieve the gold standard for scientific inference, evaluations of a tool's ecological effectiveness should aim to avoid bias by randomly assigning control and treatment groups and consistently implementing interventions across all groups. An evidence-based, case-control study should thus ideally involve a comparison between a randomly selected treatment livestock herd that is exposed to an intervention and a control livestock herd that is not exposed. While the reviews acknowledge that a tool's effectiveness is context dependent and subject to complex ecological and social confounds, they nevertheless urge ecological evaluations to use measurements of effectiveness that are as controlled and unbiased as possible. All reviews therefore excluded correlational studies or looked at them only as a supplement to their analysis. van Eeden et al. (2018) also acknowledged that input from multiple stakeholders, including scientists and livestock producers, are needed to guide the empirical tests of tools and contribute to the research process.

Like these studies, we too defined ecological effectiveness as a change in carnivore behavior (i.e., detections) within an experimental framework. The relationship between detections and predation is complex. In another study of Foxlight effectiveness, Naha et al. (2020) found that Foxlights led to a significant decline in livestock predation but no difference in leopard visitation between experimental and control sites. Thus, deterrents may diminish a carnivore's willingness to expend the energy and assume the risk associated with predation without altering visitation rates (Wilkinson et al., 2020). Nevertheless, it is important to look at detections in addition to predation events because the harassment and stress associated with mere carnivore presence can affect the health of livestock herds (Ramler et al., 2014).

Taken together, ecological effectiveness and social acceptability contribute to the social effectiveness of a given tool and determine adoption. When analyzing Foxlights, we acknowledged that the social effectiveness of a tool varies across individuals, systems, and timescales. We use this study as an example of how taking these considerations into account can improve future evaluations of tools like Foxlights.

METHODS

We organized this methods section to reflect the approaches we took to analyze both social acceptability and ecological effectiveness. We first discuss one, then the other. We took our pre-understanding into account before beginning this process and recognized that our analysis of the data would mirror our individual backgrounds and contextual knowledge. Our group of coauthors have a uniquely interdisciplinary background steeped in social and ecological science, and we have conducted research at the study site (HREC) in some form since 2014. This granted us familiarity with the California rangeland system and with the local dynamics of conflict throughout interviews. We were always transparent about our backgrounds with producers and made it clear that our goal was to thoroughly integrate producers into the research process.

Producer Attitudes

Interviews

We conducted semi-structured interviews with 11 sheep and cattle producers in Northern California before and after completing our empirical evaluation of Foxlights (see section Predation Deterrent Experiment). Given our qualitative approach, our interviewee pool was small and the results were not intended to have universal applicability or be generalized statistically. These livestock producers operated in Mendocino, Alameda, Sonoma, San Mateo, and Contra Costa counties (all of which are geographically, climatically, and culturally similar), while the Foxlight experiment took place at the Hopland Research and Extension Center (HREC) in Mendocino County. This region has a history of sheep production, with a decline in recent decades. Many producers attribute the decline to an increase in coyote predation (Larson et al., 2016), while other sources attribute it to broader economic change (Berger, 2006).

We began by interviewing HREC producers who managed the sheep flocks that were involved in the Foxlight experiment. These producers had a professional stake in HREC's sheep management and some input on the sheep program but no direct stake in the finances of the program. Other interviewees were then identified via a network sampling technique, which involved contacting future interviewees from recommendations of past interviewees (Noy, 2008). The only requirements were that the interviewees identified as livestock producers and were willing to be interviewed. Of the 11 interviewees, three were employed at HREC, three operated on privately leased lands, three on publicly leased lands, and two on a mixture of public and private

leases. Livestock herd size ranged from one producer who was responsible for 70 sheep to another producer who ran 600 mother cows. All 11 producers had experienced livestock loss to coyotes, ranging from one producer who stated that they lose 25% of their calf crop to coyotes every year to another producer who only had one experience with coyote predation.

Each interview lasted from 30 min to 2 h. We started with a set of predetermined open-ended questions (**Supplementary Material**) and posed additional questions as the conversations evolved. Interviews covered tool use, information sources, identity and landscape change, definitions of coexistence, affect toward carnivores, and the material and emotional costs associated with livestock loss. All interviews were recorded with permission from the interviewees and transcribed for analyses. The study protocol was reviewed and approved by the University of California, Berkeley Committee for Protection of Human Subjects (CPHS Protocol Number: 2019-02-11801).

After the Foxlight experiment analyses were completed, we contacted the 11 previously interviewed producers to investigate whether empirical findings would change their attitudes toward Foxlights. This second round of interviews involved briefing interviewees on the study results without firm claims on the conclusions. We began by explaining the research and our desire to use a SES approach to evaluate a livestock protection tool that incorporated producer perspectives. We then presented the Foxlight experiment methods and results with special attention to the lack of a strong signal in the data. We made it clear that our study was not able to make any definitive claims about how Foxlight presence interacted with predation due to limited data. Interviewees asked clarifying questions throughout the presentation and sometimes proffered their own interpretation of the empirical results. To ensure continuity, the same authors who conducted the interviews also transcribed and analyzed interview transcripts.

Interview Analyses

We employed a qualitative content analysis method known as *manifest analysis* (Elo and Kyngäs, 2008; Bengtsson, 2016; Carlson, 2018; Okumah et al., 2020; Pimid et al., 2020) to examine interview transcripts. This method emphasizes staying close to the original data and is unique because it has both quantitative and qualitative methodology.

Each transcript was analyzed through hand coding (**Figure 1**). The first stage was *decontextualization*, where we began with a precursory reading of the transcripts, followed by a second readthrough where certain quotes were selected and color-coded by theme (Bengtsson, 2016). Examples of themes included "definitions of coexistence" or "opinions toward science. "Selected quotes were paraphrased into meaning units by cutting crutch words or redundant phrases while staying true to the text. After this, meaning units were assigned codes that we created throughout the analysis process. We used *inductive content analysis* to create codes based on abstraction from the specific to the more general while remaining as text-driven as possible (Elo and Kyngäs, 2008; Graneheim et al., 2017). We relied on *open coding*, a unique component of inductive content analysis, to detect patterns and freely generate categories as we read and re-read transcripts. This allowed us to imbue the original text with agency but also meant that codes evolved as the study progressed. To avoid obscuring the meaning of these codes, the coding process was repeated until codes stopped evolving, which often involved collapsing similar but more specifically worded codes into broader and more generalizable versions.

After assigning codes, we began the "compilation stage," where we combined a quantitative and qualitative approach to detect patterns and extract meaning from the text. We counted the number of times a given code appeared across all interviews and presented the final number in the tables. Even though a single code could be present multiple times within a single transcript, codes were only counted once per interview. Then we progressed to the writing process, where we used manifest analysis to gather meaning from the text, which is what is presented in the results. Manifest analysis involves describing *what* the informants say, as opposed to trying to find hidden meanings or subtext. Thus, we referred to the original text as much as possible. Together, these quantitative and qualitative approaches helped us conceptualize social effectiveness by allowing us to assess how the various elements of social acceptability interacted with the demonstrated ecological effectiveness of Foxlights.

Predation Deterrent Experiment

Study Site

The Foxlight experiment took place at the Hopland Research and Extension Center (HREC) in Mendocino County. HREC is a 5,358-acre sheep production and education facility in the Mayacamas Mountains. University of California acquired the study site, a former sheep ranch, in 1951, and has been managing sheep on the site ever since. Native carnivores at the site include coyotes, black bears (*Ursus americanus*), mountain lions (*Puma concolor*), and bobcats (*Lynx rufus*). Coyote predation is the main issue for sheep at HREC (Scrivner et al., 1985; Neale et al., 1998; McInturff et al., 2020), with ewe and lamb loss ranging from 1 to 3% a year since 2015. Sheep at HREC are generally moved between fenced pastures every 2 to 6 weeks. Sheep flocks are most vulnerable to predation during lambing season, which occurred twice during this study from November to March. At the start of the study, the operation supported 450–500 ewes over 32 pastures, but was reduced to 135 ewes in June 2019 due to budget and staffing constraints.

Study Design

We tested the behavioral response of coyotes to Foxlights from October 2018 to January 2020 using an experimental design. We compared coyote detections between treatment sites, or camera traps in areas that were in the line-of-sight of a Foxlight (henceforth active Foxlight sites), and control sites, or camera traps set in areas without Foxlights (henceforth inactive Foxlights sites). We selected six pastures based on the recommendations of HREC producers, prioritizing areas that were commonly occupied by sheep flocks and/or reportedly frequented by coyotes. Five of the six pastures were used for sheep grazing at some point during this study. Each pasture contained paired camera trap sites (an active Foxlight site and inactive Foxlight site), yielding 12 total camera trap sites (**Figure 2**). Camera

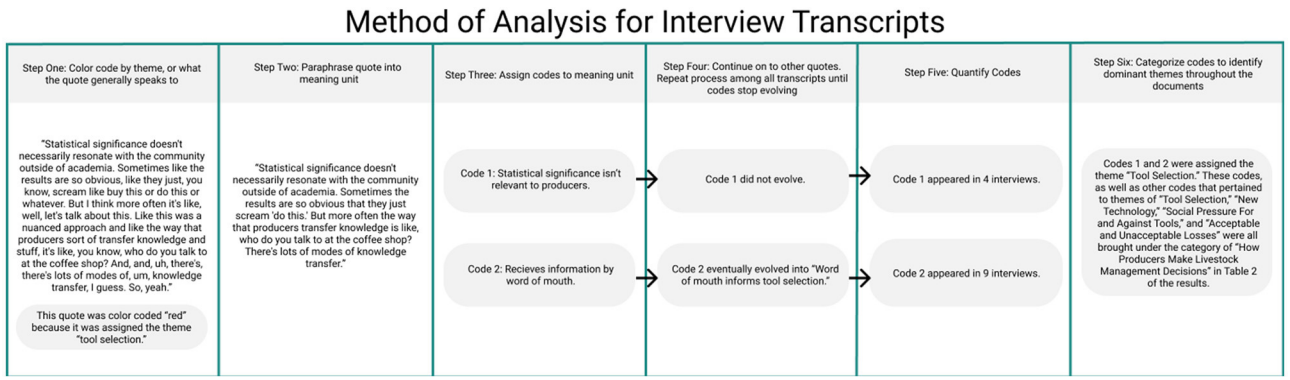


FIGURE 1 | Method of analysis for interview transcripts. This figure demonstrates how we created meaning units, derived codes from meaning units, and then quantified codes to be presented in tables.

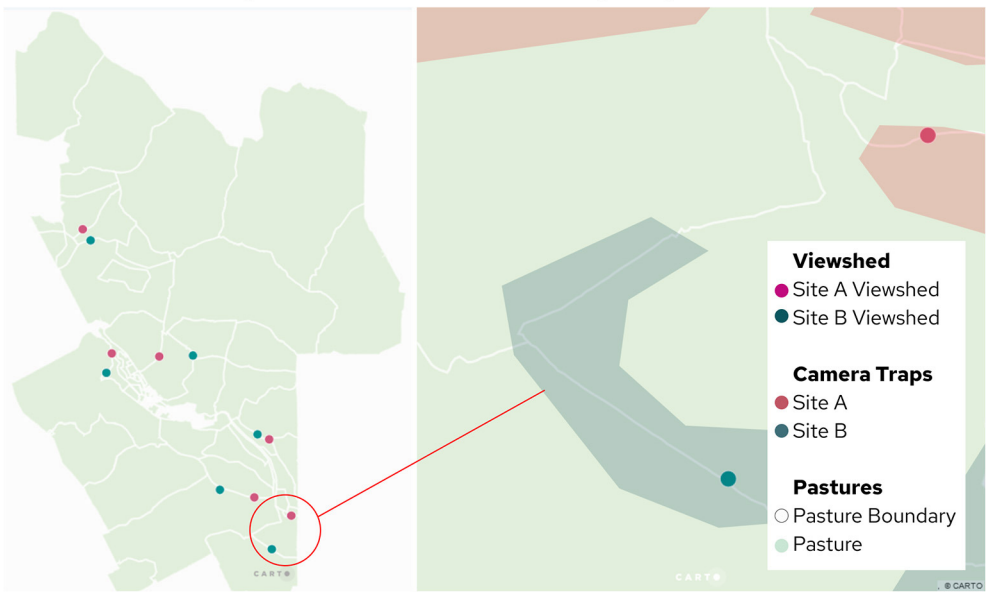


FIGURE 2 | Map of HREC with two paired camera trap sites per pasture. Each pasture's Foxlight was alternated between two sites every 5 weeks, and a camera trap was placed at each site to continuously monitor coyote activity throughout the phase. Each phase was composed of two 5 week periods (such that within a given phase, each camera trap site had 5 weeks with a Foxlight, and 5 weeks without a Foxlight). In the figure, "viewshed" represents the areas of the pasture that were in the line-of-sight of the Foxlight when it was active. It was not possible to view a Foxlight in one site from any other. Image produced on Carto.

traps were placed near coyote sign (i.e., game trails, dig holes, fence brakes) to maximize detections (Way and Eatough, 2006; DeVault et al., 2008). Camera traps (Bushnell Trophy Cam) were programmed to take bursts of two pictures at 10 s intervals when triggered and set with a normal sensor level.

Throughout the study, one Foxlight was always operational in each of the six pastures. We ensured that Foxlights were not visible from any other camera trap site, even when deployed in the same pasture. Foxlights were placed in prominent areas, such as atop of a knoll, in the center of narrow pastures, or atop of fences in larger pastures, and within 100 m from the camera trap. Within each pasture, we moved the Foxlight between the active and inactive site every 5 weeks. We defined a study "phase" as a 10-week period during which each Foxlight was active for 5 weeks and inactive for 5 weeks at a given camera trap site. There were a total of 4.5 phases during our study. Due to camera trap malfunctions, some cameras had incomplete phases whereas other cameras had longer phases. We corrected for these differences in our analysis.

Analytical Methods

We used Generalized Linear Mixed Models (GLMMs) to analyze the effects of Foxlights on coyote activity patterns. We determined the number of coyote camera trap detections during

each 5 week active or inactive Foxlight period and used this measure as the dependent variable in all models. We counted camera trap photos that occurred within 15 min of another as one independent detection, as inspection of the raw data suggested that this interval captured unique coyote groups while minimizing pseudo replication (following Šver et al., 2016; Dorning and Harris, 2019).

We used a negative binomial model to account for overdispersion of the count data and included the number of operational days in each active and inactive Foxlight period as an offset in the models to account for differences in sampling effort across camera phases. The covariates we considered to influence coyote activity were Foxlight status (binary variable), sheep presence as a potential coyote attractant (binary variable), phase in order to measure habituation (1–4), and ruggedness at a resolution of 2,500 m^2 around each individual camera trap because it was assumed to have an impact on Foxlight visibility. To ensure that correlated covariates were not confounding the results of our analyses, we tested all covariates in the top model for collinearity and confirmed that variance inflation factors (VIF) < 4 (this was the case for all models). We scaled ruggedness prior to modeling (mean of 0 and standard deviation of 1). We included camera as a random effect in all models, which controls for habitat variables. We selected the best model based on AIC (Burnham and Anderson, 2002).

We also examined whether Foxlights affected coyote diel activity patterns. To account for the circularity of the data and for seasonal differences in sunset and sunrise time, we scaled all times to radians so that $\pi/2$ corresponded to sunrise and $3\pi/2$ to sunset. We used kernel density estimation to model diel activity patterns for coyotes during periods with and without Foxlights (Ridout and Linkie, 2009). We used Watson's two-sample test of homogeneity to test for differences in daily activity patterns in areas with and without Foxlights, using the circular package in R (Agostinelli and Lund, 2017).

RESULTS

Results are organized based on social acceptability (sections Producer Attitudes Toward Foxlights Prior to Seeing Results and How Producers Make Livestock Management Decisions), ecological effectiveness (section Effect of Foxlights on Coyote Activity), and social effectiveness (sections How Producers Interpreted the Results and Attitudes Toward Science and Our Methods).

Producer Attitudes Toward Foxlights Prior to Seeing Results

Ten of the eleven interviewees utilized non-lethal deterrents or strategies. These strategies included, in order from most frequently to least frequently cited: guardian animals, human presence, electric fencing, Foxlights, night penning, strategic pasture selection, tighter calving/lambing season, E-collars, solar motion lights, fladry, and radios.

Most interviewees were either willing to use Foxlights or already used them (**Table 1**). One of these interviewees stated, "Yeah, I would [adopt Foxlights]. I would do it if somebody gives me a new idea on how to deter predators from sheep. I would use it in a second and then watch probably for a season to see if it was working and then if it worked out, keep doing it. If it didn't, I would look for something else." In contrast, interviewees that were unwilling to try Foxlights either had concerns about the feasibility of deploying deterrents on public land, had too few issues with predation to warrant investing in deterrents, or did not believe in the effectiveness of non-lethal deterrents. To this latter end, one interviewee stated, "I know a lot people don't like to hear this, particularly in the academia world, but the only effective way to control, particularly the coyotes, is lethally. I'm familiar with the system [Foxlights] that you mentioned, but they're just not feasible."

TABLE 1 | Producer attitudes toward Foxlights prior to seeing results of the experiment.

Theme	Code	Interviewees
Attitude toward Foxlights	No prior knowledge of Foxlights	4
	Willing to adopt Foxlights	4
	Already uses Foxlights	4
	Not willing to adopt Foxlights	3
Concerns about Foxlights	Lethal means are the best strategy against predation	4
	Concerns about habituation	3
	Concerns about lack of feasibility on public lands	3
	Concerns about cost	2
	Concerns about ruggedness and terrain	1

Each quote, or meaning unit, was assigned a code that corresponded with a theme. Meaning units could have multiple codes. The codes presented here are derived from interview transcriptions of 11 interviews. Interviewees are color-coded by percentage–a dark box color means more interviewees had meaning units in their interviews that were assigned that code than a light box color.

How Producers Make Livestock Management Decisions

According to both rounds of interviews, interviewees relied on multiple outlets and factors to make ranch management decisions (**Table 2**), with word-of-mouth serving as the most prominent information source. Tool adoption, as one interviewee described it, "depends on who recommends that tool." Producers who identified word-of-mouth as an influence on their decision-making described various kinds of relationships, listed here from order of most frequently to least frequently cited: other producers, neighbors, landowners, suppliers, friends, and researchers. When producers did get information from researchers, the researchers often either worked for their land management agency or had worked with a producer they personally knew. Producers did not commonly rely on academic research papers to make decisions, as one interviewee stated, "I'm certainly not combing through research journals as a

TABLE 2 | How producers make livestock management decisions.

Subject	Code	Interviewees
Tool selection	Word of mouth informs tool selection	9
	Other producers inform tool selection	7
	Tradeshows, workshops, and seminars inform tool selection	6
	Agricultural websites, newsletters, and catalogs inform tool selection	5
	Landowner informs tool selection	4
	Personal experience informs tool selection	4
	Scientific papers inform tool selection	4
	Suppliers/manufacturers inform tool selection	4
	Wildlife organizations inform tool selection	3
	Does not seek information when it comes to selecting tools	2
New technology	Livestock producers need new technology	6
	Livestock producers do not necessarily need new technology	2
Social pressure for and against tools	Stated personal preference for either lethal or non-lethal tools	9
	There is social pressure from the public against lethal tools	9
	There is social pressure from other producers to use lethal tools	5
	There is social pressure from other producers against lethal tools	2
	There is social pressure from other producers to use non-lethal tools	2
	There is social pressure from other producers against non-lethal tools	1
Acceptable and unacceptable losses	There is a such thing as an acceptable level of loss to carnivores	9
	There is no such thing as an acceptable loss to carnivores	1
	Not all coyotes pose risks, only certain "problem" coyotes do	9
	Carnivores can display acceptable or unacceptable behavior	7

Each quote, or meaning unit, was assigned a code that corresponded with a theme. Meaning units could have multiple codes. The codes presented here are derived from interview transcriptions of 11 interviews. Interviewees are color-coded by percentage–a dark box color means more interviewees had meaning units in their interviews that were assigned that code than a light box color.

producer." Five producers also mentioned that their access to tools, including lethal strategies, was limited by the sites or conditions of where and with whom they worked.

Interviewees cited social pressure and personal preference as influencing tool selection. For example, one interviewee stated that social pressure "can go both ways. There's social pressure to adopt non-lethal and there's social pressure from the other side to adopt lethal. Because any coyote I take out isn't predating my neighbors. So when you have a core group of say four or five ranches that are all bordering each other, they're going to put pressure, you know, I'm doing my part to get rid of the coyotes, what are you doing? But definitely there's more pressure to do the non-lethal stuff than there is anything else." Another interviewee stated, "I would say, lethal control aside, I don't think there is any public pressure on one tool vs. another. It's like, if something worked for you use it. If it doesn't work for you, don't use it. You know, try it out, let me know how it works."

Several interviewees discussed their approach to deciding between non-lethal and lethal strategies for managing conflict, displaying varying thresholds of tolerance for livestock loss or carnivore behaviors before implementing lethal strategies. For example, one interviewee stated, "I find [coyotes] really interesting and exciting, but there's this threshold that's crossed if they're inflicting damage to my animals." Examples of unacceptable livestock loss included: more than 1% of cattle a year (depending on how many preventative measures were in place), more than 2% of ewes, more than one or two ewes, losing multiple animals in a short period of time, or if all livestock losses occurred within one herd. Unacceptable behaviors included: when carnivores were particularly wasteful (i.e., mass predation events or if carnivores only ate a small part of an animal), when carnivores "packed up" into large numbers, when carnivores demonstrated habituated behavior (i.e., lack of fear of humans), when carnivores predated healthy animals as opposed to weaker ones, or when carnivores entered atypical areas.

Effect of Foxlights on Coyote Activity

Our experimental evaluation of Foxlights at the Hopland Research and Extension Center (HREC) recorded a total of 305 coyote detections over 4,915 camera trap-nights. The mean number of coyote detections per active Foxlight period at a given camera (5 weeks) was 2.1 (SD +/−3.15; **Figure 3**). For inactive Foxlight periods, the mean was 2.45 (SD +/−3.1; **Figure 3**).

None of the models of coyote activity that we tested improved upon the null model (**Table 3**), though four models were within 2 delta AIC of the null model and one model had the same AIC as the null model (Model 1). Therefore, they may all be considered top models. Model 1, which included Foxlight status (coefficient estimate = −0.12, SD = 0.22), ruggedness (estimate = −0.61, SD = 0.26), and an interaction between the two (estimate = 0.25, SD = 0.26), suggested that ruggedness reduced the impact of Foxlights on coyote activity. Model 4, which only

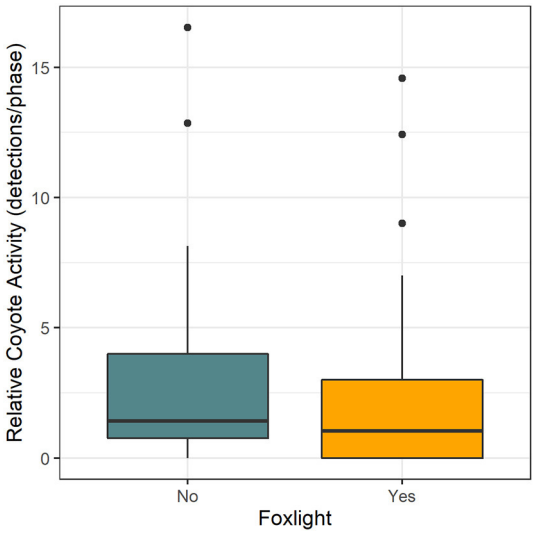


FIGURE 3 | Relative coyote activity by Foxlight status. Relative coyote activity represents the number of coyote detections in a given 5 week period.

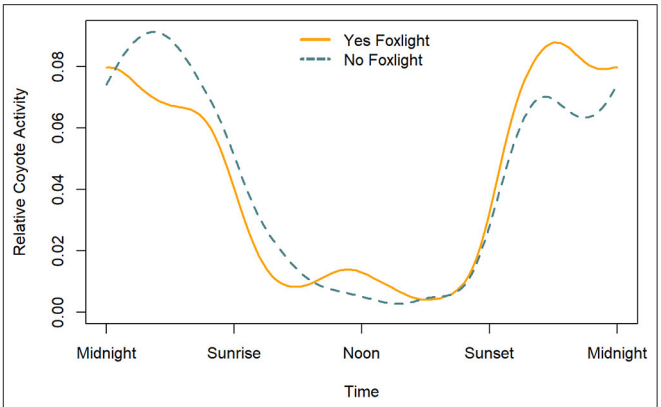


FIGURE 4 | Relative coyote activity by time of day. The lines represent the density of 24-h coyote diel activity across all periods with active Foxlights and inactive Foxlights.

TABLE 3 | Model selection for coyote detections across Foxlight phases.

Model ID	Covariates	AIC	ΔAIC
Null	**Null: (1|Camera)**	**424.4**	**0**
Model 1	**Model 1: Foxlight * Ruggedness at 2,500 m^2**	**424.4**	**0**
Model 2	*Sheep*	*424.7*	*0.3*
Model 3	*Phase*	*424.8*	*0.4*
Model 4	*Foxlight*	*425.6*	*1.2*
Model 5	*Foxlight + Sheep + Phase*	*426.4*	*2*
Model 6	Foxlight * Sheep	427.5	3.1
Model 7	Foxlight * Phase	427.7	3.2

Table includes all candidate models, including the null model (intercept and random effect only). The best models (lowest AIC) are bolded. Other candidate models within 2 AIC of the best model are italicized. AIC, Akaike Information Criterion. ΔAIC, difference between model AIC and the AIC of the best model.

included Foxlight status as a predictor, suggested that coyote activity decreased when Foxlights were active (Model 4, estimate $= -0.20$, SD $= 0.22$). Other top models suggested that coyote activity increased when sheep were present (Model 2, Model 5), and generally decreased with phase (Model 3, Model 5). There was no evidence to suggest that an interaction between Foxlight status and sheep (Model 6) or Foxlight status and phase (Model 7) influenced coyote detections.

There was no difference in diel activity patterns of coyotes at sites with active Foxlights than at sites without active Foxlights (**Figure 4**, Watson's $U^2 = 0.098$, $p > 0.10$).

Coyotes predated 14 sheep during the span of the study on the nearly 5,400 acres of HREC property. Of these 14 deaths, 6 occurred in pastures that were in our study area, and none occurred in the line-of-sight of an active Foxlight.

How Producers Interpreted the Results

Nine of the original eleven interviewees agreed to a second interview (**Table 4**). After being briefed on results of the Foxlight field study, eight of the nine interviewees either stated that Foxlights were effective or that Foxlights had the potential to be effective. For example, one interviewee stated, "It seems like Foxlights are not as effective as we would like them to be. But most of us know that this is not the only tool and anything that helps even a little bit is probably worth trying." Another interviewee stated that, "There's a good chance that with more precise, timely usage that [Foxlights] would be more effective. My feeling is that I'd probably be better at using them than they were used. So [your study] leads me to err on the side of using them, which I ultimately think what applied science is about." Interviewees that were already willing to adopt Foxlights or were already using Foxlights in the first round of interviews retained their stance on the ecological effectiveness of Foxlights after viewing our results. However, two of the three interviewees that had been unwilling to adopt Foxlights stated that Foxlights had the potential to be ecologically effective after viewing the results. The third interviewee retained their stance that Foxlights are ecologically ineffective.

When asked what our study may have overlooked, eight of the nine interviewees emphasized taking the natural histories of coyotes into account, timing the use of deterrents with seasonal changes in their activity and behavior, and identifying what other landscape variables may or may not push coyotes to undertake the risks associated with sheep predation (e.g., two interviewees postulated that if lethal take had recently fractured coyote social dynamics, coyotes on the site may have been less risk averse). Recommendations included looking at how coyotes change their behaviors based on: time of year, drought conditions, prey populations, pupping, the activities of neighboring livestock operations, and calving/lambing season. Interviewees emphasized holism, system dynamics, and context. For example, one interviewee stated, "Because of the system dynamics, even if only one out of ten coyotes is afraid of a

TABLE 4 | How producers interpreted the results.

Theme	Code	Interviewees
Foxlight efficacy	Foxlights are effective	4
	Foxlights have the potential to be effective if used differently than in this study	4
	Foxlights are not effective	1
Recommendations for foxlight use	Foxlights would be more effective if used in flatter areas	4
	Foxlights should be studied across multiple operations	3
	Foxlights would be more effective if used in more targeted areas	3
	Foxlight placement should be more randomized than it was in this study	3
	Only use Foxlights during lambing or calving season	3
	Use more Foxlights than what were used in this study	2
Recommendations for research	Environmental factors and coyote behaviors should be incorporated into analysis	9
	Cares more about predation events than coyote detections	5
	There is value in analyzing multiple deterrents at once	5
	There is value in analyzing individual deterrents	4
	This research did not overlook anything.	3

Each quote, or meaning unit, was assigned a code that corresponded with a theme. Meaning units could have multiple codes. The codes presented here are derived from interview transcriptions of 9 interviews. Interviewees are color-coded by percentage–a dark box color means more interviewees had meaning units in their interviews that were assigned that code than a light box color.

Foxlight, that means I get one more lamb a year, maybe, and I've paid for my Foxlight, right?" Another interviewee stated, "On a flat field with no terrain to speak of, maybe [Foxlights] would work. But I'm not unconvinced to buy one. I would still try it. Context in general [is my biggest consideration]. There are so many other variables that you can't control for in research and especially in rangelands. All of [the other factors] are things that still make me want to try a Foxlight." A third interviewee stated that Foxlights are "not that effective. In the context that they were tested in. I still feel like they would be effective in a different context, but it makes sense to me why it wouldn't have been that effective in the broad acreage."

Five of the nine interviewees stated there is value in analyzing multiple deterrents at once. For example, one interviewee stated that they would "love to see a chart that's like—what are the most effective tools in combination." Another interviewee stated that if the goal is to "try to prevent coyote predation of sheep and it doesn't matter what tools you use, then you would maybe do a study on a combination of tools [to see] what works best." Other interviewees warned against too much complexity. To this end, one interviewee stated that "sometimes holism and complexity can be an excuse to arrive at a point where you kind of give up on actual decision-making."

Attitudes Toward Science and Our Methods

Interviewees expressed opinions toward science in the context of rangeland management throughout the course of the two interviews (**Table 5**). Half of the interviewees stated that science can be biased, three stated that personally trusting or knowing the researchers is what makes science significant to them, and only one interviewee stated that producers are the intended audience of livestock-carnivore conflict research. Otherwise, interviewees tended to identify "other researchers," "policymakers," or "customers" as the target audience of research. Five interviewees stated that they trusted the validity of this study after viewing the empirical results, citing its lack of bias, its systems-oriented approach to methods and analysis, its inconclusive results, its accessible explanation of the results in "layman's terms," and its incorporation of producer perspectives. For example, one interviewee stated, "There are types of research that seem really aware where you interview producers, like this is great that you're interviewing producers and I think that really feels valuable to me [because it] makes it seem like this is actually applicable." As a demonstration of perceptions of bias, another interviewee stated, "I judge research by the people that do it, and there are very few people I trust doing livestock research. [This study represents] a group I got to know and I trust them. They had no personal agenda involved, and that's key." As for perceptions of exclusion, another interviewee stated, "Like as a producer, [we] would look at [scientific papers] and say, this is specifically written so I cannot understand it. You know, to make it exclusionary or whatever. So maybe that's why producers wouldn't read that. Not because they're not interested, but because it's just too academic in a different perspective, almost in a different language. I think if a lot of these results were put out in a more usable, friendly format to people, they would for sure pay attention."

DISCUSSION

Our research demonstrated that an integrated assessment of social effectiveness that combines ecological effectiveness and social acceptability adds critical new dimensions to our understanding of the broader capabilities and adoption of non-lethal livestock protection tools.

Our empirical results provided weak evidence that Foxlights affect coyote activity, but most livestock producers we

TABLE 5 | Attitudes toward science and our methods.

Subject	Code	Interviewees
Attitudes toward science	Referred to at least one research paper or study when discussing carnivores	6
	Science can be biased	5
	Trusts the validity of this study despite perceptions of bias in science	5
What makes research significant	Research is significant when it aids with decision making	5
	Statistical significance is not relevant to producers	4
	Statistical significance is relevant to producers	3
	Research is significant when researchers are trusted	3
	Research is significant when it uses real-life ranching operations	3
	Research is significant when it is presented in a way that is accessible to producers	2
Attitudes toward our study	Trusts our research because they agree with and understood our methods	5
	Trusts our research because we did not have a personal agenda	4
	Trusts our research because our results were inconclusive	1
Why they agreed to be interviewed	Wants to contribute to the progression of rangeland science	4
	Wants to know if tools work	3
	Knows and trusts the interviewee	3
	Research like this brings two different perspectives together	2
	Suspected their perspective was not being represented	2

Each quote, or meaning unit, was assigned a code that corresponded with a theme. Meaning units could have multiple codes. The codes presented here are derived from interview transcriptions of 11 interviews. Interviewees are color-coded by percentage–a dark box color means more interviewees had meaning units in their interviews that were assigned that code than a light box color.

interviewed still believed that Foxlights had the potential to be effective in conjunction with other strategies. Thus, the field of HWC would benefit from broadening established definitions of ecological effectiveness to include critical but often overlooked components of social acceptability, knowledge transfer, and dynamic socio-ecological systems. Researchers need to be aware that the social acceptability of a tool as well as systems-oriented approaches to tool evaluation are particularly relevant to stakeholder goals and perspectives when communicating science, and they should not expect the ecological success or failure of a given tool to be persuasive to a producer that is accustomed to working with complex systems in their husbandry. While our small sample size (11 interviewees) limits the universal applicability of our findings, the process by which we attained our results sheds light on how iterative collaboration can foster trust for research and promote goodwill between stakeholders. Our research also serves as a model for how a transdisciplinary approach can help future studies incorporate both social acceptability and ecological effectiveness into their methods of analysis.

The Value of a SES Approach to Tool Evaluation

Prior to learning the results of the Foxlight experiment, producers generally had an attitude of "anything helps." After we showed them the weak empirical results of the Foxlight experiment, interviewees in the second round of interviews still tended to believe that Foxlights had the potential to be effective. They acknowledged that deterrent effectiveness can be influenced by context (Eklund et al., 2017) and recognized that deterrents often work in association with each other to create an overall impact. It was clear that interviewees did not expect Foxlights to replace their preexisting strategies or even expect Foxlights to always work, likely because they recognized how environmental variability can impact an individual tool's ecological effectiveness. It is also possible that interviewees were more willing to think of Foxlights as effective because no sheep loss to coyotes occurred while in the line-of-sight of Foxlights over the course of the experiment, even though we clarified during interviews that low sheep mortality throughout the study period limited our ability to examine the effects of Foxlights on sheep predation. When asked about the results of our experiment, interviewees tended to focus on brainstorming new ways use the tool effectively instead of concentrating on the deficiencies of Foxlights. In other words, our empirical analysis did not give them reason to dismiss Foxlights as ineffective, but rather it gave them reason to lean into finding ways to make it more effective. Thus, empirical examples of effectiveness may not be what drives producer attitudes toward tools like Foxlights.

When it came to suggestions for different approaches to studying tools like Foxlights, interviewees tended to recommend approaches that reflected SES principles. They emphasized the importance of incorporating environmental variability, coyote ecology, and other management strategies into empirical evaluations of tools. The producers that we interviewed specifically identified that the established definition of ecological effectiveness that we presented them—the ability of Foxlights to deter coyotes from pastures—was inconsistent with their experience and way of thinking. An experimental method of isolating and testing a tool individually was not realistic to the interviewees' practice. Instead, they thought of tools as

part of a complex and dynamic system that demanded an adaptive toolkit. This means that scientific reductionism does not always align with livestock producers' systems-oriented approaches to husbandry. We instead recommend systems-oriented evaluations of non-lethal tools, such as testing tools in combination as well as adjusting research variables to incorporate what producers identify as important. Analyzing multiple tools at once may enable producers to cycle through tools throughout the year, thus only applying tools when they can be most effective and avoiding habituation. Two interviewees also speculated that using Foxlights in combination with other tools and strategies would further allow coyotes to expect the association between risk and light through a process of "sensitization" (Blumstein, 2016; Gaynor et al., 2020).

Whether using multiple tools to sensitize carnivores or prevent habituation, few studies have examined multiple tools at once, but those that have offer promising results (Espuno et al., 2004; Lance et al., 2010; Garrote et al., 2015; Manoa and Mwuara, 2016; Miller et al., 2016; Moreira-Arce et al., 2018). For example, the Wood River Wolf Project used a range of non-lethal strategies and deterrents, including Foxlights, to lower sheep predation at sites in Idaho by 90% (Stone et al., 2017). The project operated on the assumption that no single deterrent can effectively reduce conflict, and the results revealed that tools must be consistently rotated, adapted to an ever-changing context, and analyzed holistically (Martin, 2020). Producers at HREC were actively employing other strategies throughout our study, something that potentially served as a confounding factor in this experiment, and our results would have benefitted from analyzing Foxlights in combination with other techniques (see **Supplementary Materials** for further discussion on the empirical results).

Research variables in future evaluations of non-lethal tools should better incorporate both the environmental and social factors that producers identify as important. Our study supports previous research that has found misalignment between producer perspectives on effectiveness and empirical analyses (Lance et al., 2010; Teague et al., 2013; Ohrens et al., 2019b). Management efforts should focus on bridging these domains of scientific and producer knowledge to inform decision-making. For example, another study involving HREC producers and their perceptions of risk demonstrated how the integration of producer perspectives into empirical assessments was essential to understanding coyote activity and deterrent use across a livestock operation (McInturff et al., 2020). We suggest that researchers select response variables that are informed by the interests of the stakeholders, not just what researchers can, or choose to, measure.

The Role of Social Effectiveness in Producer Decision Making

Our finding that ecological effectiveness alone is not enough to alter producers' attitudes builds off the work of Brunson (1992), who revealed how an overreliance on technical information can be detrimental to social acceptability for multiple reasons that our findings support, including: stakeholders are often already educated on the technical aspects of a subject, scientific jargon can alienate producers, overreliance on one "right answer" can fail to account for environmental heterogeneity, and that science cannot resolve differences of opinions that correspond with belief systems. Furthermore, livestock producers make decisions through holistic considerations of production dynamics by relying on both technical and cultural knowledge transfer. For example, a producer may learn about system dynamics from older generations of ranchers, their own experience of their land, and from scientific sources. Most importantly, producers *intentionally* engage with diverse knowledge sources when it comes to understanding the socio-ecological systems they operate within (Wilmer and Fernández-Giménez, 2015). The fact that scientific demonstrations of a tool's ecological effectiveness serve as only one source of information among many for producers underscores the need to incorporate social acceptability into tool evaluations.

Several elements that contribute to social acceptability were brought up in interviews. Interviewees emphasized that the messenger of scientific findings is important because there must be trust in who recommends a tool (Section Social Trust). In our study site, as for much of the American West, social trust between agricultural producers and scientists is low (Bonnie et al., 2020). Over half of the interviewees held negative attitudes toward science, which perhaps explains why other producers often serve as their most reputable source of information. But after working with us through multiple rounds of interviews, producers began coming to us for more information and discussion, demonstrating that research itself can build social trust if stakeholder perspectives are meaningfully included in the process. For example, all three of interviewees that worked at HREC expressed their suspicion of bias in science. Yet all three were among the interviewees who stated that they trusted the validity of our research, perhaps because they either played a role in the design of the experiment's methods, witnessed the research onsite, or like the other interviewees, participated in iterative interviews. When working with producers, researchers need to acknowledge their own positionality and account for the various ways they may be perceived. Investing in truly participatory science with stakeholders at multiple checkpoints throughout an experiment will both foster trust and address the perception that science or conservation can be biased or exclusionary (Hazzah et al., 2019). These findings underscore the value of stakeholder collaboration in informing social trust and social acceptability.

The role of attitudes, values, and systems in informing social acceptability also manifested in interviews. Interviewees elaborated on their personal values and identities, and how their attitudes toward lethal or non-lethal control often influenced their decision on whether to adopt certain methods (Section Attitudes and Values). They tended to express positive or negative attitudes toward the recent value shift in the American West. For example, when asked if the way rangelands are being managed is changing, one interviewee stated, "Over the course of time in California, people's emotions have taken over common sense. They let emotions drive their votes, and their votes have taken away all the effective means to control these predators. And that's the biggest frustration you have when you live in California."

Alternatively, a different interviewee on the same subject stated, "[The change] is a very emotional issue for a lot of these old guys. This is a human problem. It comes down to a sense of entitlement that this landscape should never challenge us and we should not have to coexist." Furthermore, the relevance of contexts and systems in informing social acceptability was certainly demonstrated within the interviews, as interviewees emphasized again and again the importance of systems-oriented examinations of HWC that account for available alternatives and environmental variability (Section Context and Systems).

Interviewees also described how they incorporated various ways of knowing into their decision-making process and relied on social networks (Section Information Transfer and Research Process). Our network sampling technique for contacting new interviewees may have even enabled us to access this knowledge network throughout our research process. As one interviewee stated, "There are people on the leading edge who are reaching out to other places and publications and are choosing [tools] they want to try. And then maybe there's enough of those people that it becomes a critical mass and then they push back on the mainstream [means of control]. It's pretty fascinating how knowledge transfers and how ideas spread." It is possible that some of our interviewees were opinion leaders on the "leading edge" of new technologies (Lubell et al., 2013), especially those that answered the question of "Why did you agree to be interviewed?" with statements that expressed their desire to either learn about new potential solutions or contribute to the progression rangeland science. We also found that social pressure from other producers played an important role in information transfer, although pressure only seemed to act on either lethal or non-lethal strategies but not between individual tools. Most interviewees emphasized that the public does not support lethal strategies, which is consistent with other studies (Naughton-Treves et al., 2003; Wolf et al., 2017; Diaz et al., 2020). Sometimes this pressure made interviewees inclined to select non-lethal tools, as one interviewee stated, "I think we're going to lose more and more of the lethal tools. And so it's really important to develop other tools that can work." In other cases, pressure had the opposite effect, as another interviewee stated, "I feel like if you give into the gimmick [and use non-lethal tools], then it's kind of a slippery slope and you're kind of giving up your option of really doing what really should happen." And finally, not only is there implicit evidence to suggest that our integrative research process that was built around producer participation contributed to the social acceptability of this tool, but producers explicitly stated that our transdisciplinary methods increased the credibility of this project (Section Information Transfer and Research Process).

Our findings also revealed how HWC mitigation has both economic and psychological dimensions within a given "taskscape," which is a social construction of a landscape that accounts for the lives, work, and practices that imbue the material landscape with meaning (Section Attitudes and Values). It is difficult, and perhaps impossible, to address one of these dimensions without acknowledging the other. For example, when asked how much they liked coyotes, one interviewee said, "I would [rate] coyotes zero. I've just seen so much gore and violence that it ceases to even be about money. It's about suffering." Livestock losses are also often unevenly distributed in space and time, obscuring the full impact. Uncertainty, and especially chronic uncertainty, has costs of its own. To exemplify this point, another interviewee said, "I'm always in a state of paranoia about that. And that's just the life of the shepherd. I hear any coyotes and I'm just like outside in my pajamas with my flip flops, trying to figure out where the sounds are coming from." While a loss of 1–3% of sheep crop to predation at our study site is a fairly standard industry loss, HREC producers explained that every predation event is a direct income loss of anywhere from $150–$500 per animal for producers, affects their job performances, and carries an emotional toll. These findings speak to the larger point illustrated by our research—ecological or economic data aren't the only forces driving attitudes when it comes to making decisions surrounding livestock loss and predation prevention. Strictly ecological or economic interpretations of the effectiveness of livestock protection tools will miss vital human dimensions, especially regarding social acceptability.

Recommendations

We recommend that researchers adopt the same systems-oriented approaches already used by producers to both test tools and communicate findings. This may involve analyzing deterrents in concert, accounting for broader environmental factors, and incorporating research variables that influence social acceptability. Researchers should continue to test tools, but also work closely with producers to solicit feedback. Establishing lending libraries of tools and partnering with producers to collect data will allow researchers to learn from their knowledge and insight, build trust, provide exposure to tools, and lower the barriers that enable access to certain tools. In the same way that app developers use business techniques to let users trial apps and "break" them in the real world, scientists could implement a similar, iterative approach with non-lethal tools, especially given that producers quite reasonably want to experiment with tools for themselves before forming opinions (Hazzah et al., 2019). We also recommend that our integrated and participatory approach be considered not just by other researchers, but also by land managers as part of their planning cycle. Land management agencies can use this iterative process to recognize a problem, identify potential solutions from stakeholder opinion and scientific literature, and then work toward a practical solution that is scientifically robust and culturally palatable. Establishing checkpoints with stakeholders along the way will allow managers to determine which solutions have social effectiveness, both in terms of solving the problem and aligning with stakeholder values. Work like this is already underway: the Wolf Advisory Group (WAG) within the Washington Department of Fish and Wildlife guides efforts to reduce conflict between wolves and livestock by inviting stakeholders from diverse backgrounds to participate within an inclusive decision-making framework (Wiles et al., 2011). Such approaches can guide tool adoption and promote sound practices, ultimately supporting conservation as well as livestock production goals. Examining systems-oriented approaches, account for social acceptability, and enabling practitioners test things for themselves may have much higher

yields for the future of coexistence than endless science on particular tools.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article has been made available by the authors in the **Supplementary Material**. Additional data will be made available by the authors, without undue reservation, upon request.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by University of California, Berkeley Committee for Protection of Human Subjects (CPHS) and Office for Protection of Human Subjects (OPHS). The participants provided their written informed consent to participate in this study. Ethical review and approval was not required for the animal study because no animals were captured or handled, and the Foxlight experiment was non-invasive. Written informed consent was obtained from the owners for the participation of their animals in this study.

AUTHOR CONTRIBUTIONS

LV, AM, and KG contributed to the conception and design of the empirical study. LV, AM, KG, VY, and JB contributed to the conception and design of the qualitative interviews. KG, AM, and LV performed the statistical analysis. LV organized the database, conducted and analyzed interviews, and wrote the first draft of the manuscript. KG, AM, and LV performed the statistical analysis. AM, KG, VY, and JB wrote sections of the manuscript. All authors contributed to manuscript revision, read, and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.682210/full#supplementary-material

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Can Deliberative Democracy Favor a Flourishing Relationship Between Humans and Carnivores?

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There is considerable interest in improving participatory governance in decision-making processes for the conservation of biodiversity and management of conflicts between humans and wildlife. Among the various modes of participatory governance, deliberative democracy has received virtually no attention for decisions focused on conserving biodiversity. This is surprising given that deliberative democracy is an important branch of democratic theory and is associated with decision-making processes that have been successfully applied to a wide range of complicated decisions across diverse cultural settings. Moreover, deliberative democracy has several distinctive properties that would seem to make it well-suited for many conservation decisions. First, deliberative democracy is better-designed than other processes to handle cases where the object of conservation appears to be insufficiently valued by those who have the most detrimental impacts on its conservation. Second, deliberative democracy engenders a rich kind of representation and impartiality that is nearly impossible to achieve with participatory governance focused on managing conflicts among hyper-engaged stakeholders. Here, we review the principles of deliberative democracy, outline procedures for its application to carnivore conservation, and consider its likelihood to favor carnivore conservation.

Keywords: carnivore conservation, collaborative governance, deliberative democracy, environmental governance, participatory governance

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SUMMARY

Carnivores are emblematic of many challenges in conservation for being insufficiently valued to reverse their dismal and deteriorating conservation status. Carnivore conservation is also like many conservation challenges in that the costs and benefits of conservation are often experienced unequally among the members of society. These circumstances represent problems for participatory governance, which tends to court participation by those already deeply committed for or against conservation. While such engagement is appropriate, there is also a need for processes that have broader representation and are more likely to elicit non-prejudicial judgments about conservation-related policies and decisions. Here, we outline a process, known as deliberative democracy, that explicitly aims to handle these concerns. This process is well-known among political theorists and has been applied to many complicated cases. Yet, it has received little attention among those interested in decision-making and governance that pertains to the conservation of biodiversity.

INTRODUCTION

Governance and decision-making in conservation take various forms that may be characterized as being situated along a spectrum, with one end representing decisions by government officials with little citizen involvement and the other end representing decisions with extensive involvement of citizens. One end of the spectrum is sometimes referred to as top-down or autocratic. The other end of the spectrum is sometimes referred to (often interchangeably) as bottom-up, participatory, or collaborative. Cases have been made that collaborative decision-making is preferable because it is more likely to produce decisions that are more fair and durable over time (e.g., Redpath et al., 2017).

Yet, the tendency for participatory governance to yield adequate conservation is not well-understood (Koontz and Thomas, 2006; Bodin, 2017), the factors that favor and disfavor adequate community-led conservation are not fully understood (Brooks et al., 2012), and criteria for judging adequate conservation are not widely agreed upon (Vucetich et al., 2021). Because the very meaning of conservation varies among authors, there is value in specifying our usage. Specifically, we use "conservation" to refer to the maintenance and restoration of species across large portions of their native, historic range at populations densities that allow for the manifestation of their ecological functions (Soulé et al., 2003; Vucetich et al., 2006, 2018). With that framing, conservation is difficult, in part, because the costs and benefits of conservation are often experienced unequally among the members of society.

Collaborative forms of conservation decision-making come in a variety of forms, such as conflict transformation (Madden and McQuinn, 2014), collaborative learning (Daniels and Walker, 2001), community-based conservation (Otto et al., 2013), structured decision-making (Gregory et al., 2012), and governance of environmental commons (Ostrom, 1990; Wilson et al., 2013). Deliberative democracy is an especially important form of participatory governance, represents the dominant form of democratic theory among political theorists (Bächtiger et al., 2018), and has been applied to many complicated decisions administered in a wide range of cultural settings (Center for Deliberative Democracy, 2021a). However, deliberative democracy has received virtually no attention with respect to the conservation of biodiversity.

Here we explore the potential for deliberative democracy to result in adequate decisions for the conservation of carnivores. The value of doing so is most readily appreciated by highlighting two concerns that routinely arise with existing forms of collaborative governance and then examining how deliberative democracy approaches these concerns in a different manner. For broader treatments of deliberative democracy, see, for example, Bohman (1998), Dryzek (2000), and Fishkin (2018).

One Concern

This first concern is well-illustrated by the principles of governing environmental commons (Ostrom, 1990; Wilson et al., 2013), which indicate that sustainable use of a natural resource is possible through self-governance at a local scale if certain conditions hold. One condition is especially salient: the resource being considered is sufficiently valuable to those using the resource (Wilson et al., 2013). An archetypal example is a group of people who understand that overexploiting a certain species of fish will importantly diminish their well-being in the foreseeable future.

The concern with this condition for governing environmental commons is that many aspects of biodiversity are in desperate need of conservation, but are not viewed as sufficiently valuable by the people who most impact that aspect of biodiversity. Examples from WEIRD nations include:

- People, often associated with hunting and ranching in the United States, who tend not to sufficiently value the existence of wolves in their native range (Nie, 2003; Carlson et al., 2020).
- People associated with hunting red grouse in the United Kingdom, who tend not to sufficiently value the existence of raptors on their native range (Thirgood and Redpath, 2008).
- People, often associated with oil and gas exploration in the United States, who tend not to sufficiently value the existence of sage grouse on their native range (Tobias, 2019).

Examples from non-WEIRD nations include:

- People, often associated with fishing in the Amazon, Yangtse, and Ganges rivers, who tend not to sufficiently value river dolphins (e.g., Kelkar et al., 2010; Alves et al., 2012).
- People, often associated with timber harvesting in tropical forests, who tend not to sufficiently value the biodiversity that is adversely impacted (Giam, 2017).

We use the phrases "*people*" and "*sufficiently value…*" in a particular manner. The word "people" is not intended to indicate that all people place low value on particular species. Rather, we use the word to indicate that groups of people living in a geopolitical community (such as, but not limited to, a state, province or nation) often do not *collectively* place sufficient value on certain species to result in such species being adequately conserved.

Situations like those listed above are common, in part, because the cost and benefits of conservation are routinely experienced differently by different agents within a community. If the biodiversity crisis is to be stemmed, there is a need for a decision-making framework that is substantively collaborative with citizens, yet is not unduly dependent on the species of conservation concern being seen as sufficiently valuable to those who threaten the species' conservation. Particular aspects of deliberative democracy have the potential to fulfill this need.

This concern is not limited to governing environmental commons. Essentially the same concern is present across participatory forms of decision-making and manifest as the difficulty of designing participatory processes with appropriate representation, i.e., without over-representing the interests of a few who do not sufficiently value the species of conservation concern (López-Bao et al., 2017).

Highlighting the importance of insufficient valuation of species does not imply that all conservation problems are usefully

characterized (or that any conservation problem is completely characterized) as such.

A Second Concern

A second concern common to many participatory processes rises from the tendency to focus on participation by citizens who are hyper-engaged with advancing or limiting conservation. These participants tend to see their personal and social identities as deeply entwined with particular outcomes of the decision-making process. Those circumstances can become obstacles to deliberating or compromising over the decision. The decisions resulting from these processes tend to be unduly influenced by power dynamics. Some collaborative processes aim to overcome these concerns by focusing on them. An important example of such an approach is conflict transformation (Madden and McQuinn, 2014). While there is a vital need to effectively tend the interests of the hyper-engaged, there is also a need for decision-making processes that better represent the interests of broader groups of stakeholders. Again, particular aspects of deliberative democracy have the potential to fulfill this need.

The disposition of this essay is not to argue that deliberative democracy is unqualifiedly better than other forms of collaborative decision making. Rather the point is to enlarge conservation decision-makers' repertoire of decision-making processes so that the most appropriate procedures can be applied to each particular situation.

LARGE CARNIVORES

While deliberative democracy can be applied to any kind of conservation decision, its distinctive strengths may be best appreciated with decisions about biodiversity in great need of conservation, but not viewed as sufficiently valuable by the humans who most impact those aspects of biodiversity. While many aspects of biodiversity match this circumstance, large carnivore species match the circumstance particularly well. Hereafter, we use large carnivore conservation to illustrate principles of deliberative democracy, though the ideas are readily transferred to other cases.

Of the planet's large (>15 kg) carnivore species, 60% are classified as vulnerable, endangered, or critically endangered by the International Union for the Conservation of Nature (IUCN). More than three-quarters of the planet's large carnivores are in decline. And, the average loss of historic range among large carnivores is ∼50% (Wolf and Ripple, 2017). The most common threats to carnivores include high rates of human-caused mortality, degradation and loss of habitat, and depletion of prey (Macdonald, 2019).

Many humans value carnivores in various ways and for a variety of reasons, including:

- the acknowledgment of carnivores' intrinsic value (*sensu*, (Vucetich et al., 2015))
- the attribution of existence value to carnivores (*sensu*, Attfield, 1998).
- the ecosystem services that carnivores provide via the direct and indirect effects of predation (e.g., Weiss et al., 2007).
- the opportunity for non-consumptive uses, such as photography and eco-tourism (e.g., Duffield et al., 2006).
- the opportunity for consumptive uses, such as hunting.

The last of those values (hunting) is controversial. Some argue that well-designed and well-implemented hunts can favor—or are even essential for—carnivore conservation (Dickman et al., 2019). As such, hunting might be an expression of the positive valuation of predators. Other are concerned that carnivore hunting tends to be an expression of the negative valuation of carnivores (Downes, 2013; Vucetich et al., 2017; Chicago Tribune, 2021), and that hunting too often works against conservation due to the apparent difficulty of reliably implementing a well-designed hunt (Creel et al., 2016, see also Vucetich et al., 2019). Regardless of concerns associated with valuing carnivores for the opportunity to hunt them (or other consumptive valuations, e.g., Coals et al., 2019a,b), the sum total positive valuation of carnivores has been insufficient to reverse their dismal and deteriorating state.

The positive valuation of carnivores is more than offset by two competing valuations of carnivores. First, some humans intensely disvalue carnivores for various reasons, including real or perceived threats to human safety, threats to livestock, and competition for opportunities to hunt wild ungulates (Vucetich and Macdonald, 2017). Second, many humans do not disvalue carnivores, but place greater positive valuation on endeavors that compete with carnivores. One of many such examples is the advocates for palm oil plantations, given the adverse impact of that endeavor on the habitat of several species of endangered felids (Macdonald et al., 2018).

In summary, carnivores represent the general circumstance outlined in the *Introduction*. That is, carnivores are among the aspects of biodiversity in desperate need of conservation, but not sufficiently valuable to the humans who most affect their conservation.

Focusing participatory governance on carnivore conservation is also apropos because large carnivores tend to engender considerable emotional valence (both positive and negative) among already-engaged stakeholders in a manner likely to compound the difficulty of deliberative decision-making (Slagle et al., 2012; Flykt et al., 2013). Finally, another reason to focus on carnivores is that they are also often umbrella species—meaning that effective conservation of carnivores often leads to the effective conservation of many other kinds of biodiversity (Burnham et al., 2012).

DELIBERATIVE DEMOCRACY AND MINI-PUBLICS

Deliberative democracy is an important branch of democratic theory (Bächtiger et al., 2018). There is no precise, compact definition for deliberative democracy. Rather, it is a diffuse set of ideas centering around the idea that deliberation by a representative set of citizens is essential for good governance. Many models of deliberative democracy include a substantive role for a deliberative mini-public (Setälä and Smith, 2018). What distinguishes most forms of collaborative decision-making from

deliberative democracy are mini-publics—both their constitution and mode of conduct. For now, it suffices to consider a mini-pubic to be a representative group of deliberating citizens. In this essay, we focus on mini-publics, what they are, how they operate, and what challenges to governance they aim to overcome.

In particular, public decisions benefit from public deliberation—i.e., impartial and informed weighing of reasons for and against various choices. Yet, not all members of a large community can substantively deliberate all the public decisions that merit deliberation. Consequently, deliberation might be held by various bodies within a community—committees of legislators, panels of technocrats charged by an executive branch of government, or a mini-public of representative citizens.

Deliberation by legislators is important; but is often compromised by partisanship, tribalism and deference to special interests, all of which are often associated with undue interest in being reelected. Deliberation by technocrats is important, but lacks sufficient legitimacy when a decision depends on normative (non-empirical) considerations that vary or remain unsettled across the community at large or when technocrats do not share the values of constituents (e.g., Evans and Hargittai, 2020).

A critical and distinctive feature of a mini-public is the process by which members are selected, and the consequences of that selection process on the quality of deliberation with respect to representativeness and impartiality. The details of selecting members of a mini-public are important and outlined below. But there is heuristic value with a slightly over-simplified notion that members of a mini-public are selected at random from the population of citizens. If the mini-public is selected with reasonable care, its properties include:

- If the mini-public is large enough it will, by statistical laws of sampling, be *representative* in two regards, as the idea applies to democratic theory and with respect to the range of normative views that characterize the larger population (Dryzek, 2000, p. 172; Goodin, 2004; Brown, 2018). See below for caveats.
- Members of a mini-public are far less likely to have been hyper-engaged in the subject matter of the decision, thereby greatly reducing members' tendency to pre-judge the decision, leading to their being more open to deliberation (Polletta and Gardner, 2018; Strandberg et al., 2019).
- Members of a mini-public are likely under-informed about salient facts and consequences of the decision. A critical part of the deliberative process is for the mini-public to become sufficiently informed. Again, details below.

These properties of a mini-public are not theoretical or merely aspirational. Extensive research demonstrate that: a properly assembled mini-public is importantly representative (Brown, 2018), many members of a mini-public tend to exhibit reasonable levels of deliberation (e.g., Himmelroos, 2017; Gerber et al., 2018), deliberation tends to lessen polarization and dogmatism (e.g., Grönlund et al., 2015; Herne et al., 2019), well-implemented deliberation tends to be associated with the adequate acquisition of salient facts (e.g., Barabas, 2004; Andersen and Hansen, 2007; Farrar et al., 2010; Esterling et al., 2011), and members often to change their minds in response to deliberation (e.g., Setälä et al., 2010; Himmelroos and Christensen, 2014). No less important, an adequately implemented mini-public includes the assessment and documentation of these features for that mini-public (e.g., Isernia and Fishkin, 2014).

The distinctive capacities of a mini-public are not a reason to dissuade genuine deliberation among legislators or technocrats, nor is it a reason to leave unattended the deep or volatile conflicts among hyper-engaged stakeholders. Rather, the capacities of a mini-public make it another essential facet of good decision-making.

The collective views of a mini-public that has deliberated on an issue are typically not binding on the authorities charged with making a decision. Rather, their views tend to be advisory, as is common to many participatory processes. The mini-public's views are also publicly available—which can aid in circumstances where the insights of the mini-public were not given due consideration or influence by decision-makers.

Deliberative democracy has been implemented with a wide range of cultures, spanning six continents (Center for Deliberative Democracy, 2021a), including cultures with low education and rates of literacy (Fishkin et al., 2017) and cultures not particularly well-characterized as democratic (Fishkin et al., 2010). Deliberative democracy has also been applied to a wide range of cases, from local decisions about urban planning (Beauvais and Warren, 2019) to multi-national issues, such as climate change mitigation and immigration policy for the European Union (Isernia and Fishkin, 2014).

As with many forms of collaborative decision-making, positive outcomes depend on numerous critical, and sometimes seemingly nuanced, details. We address many of these details in the sections that follow. The following sections are organized according to these topics:

- Specifying the subjects of deliberation for carnivore conservation
- The product of deliberation
- Selecting members of a mini-public
- Enabling deliberators' knowledge
- The venue for and structure of deliberation
- Pre- and post- deliberation surveys
- The legitimacy of a mini-public
- Is deliberative democracy likely to favor carnivore conservation?

THE SUBJECT OF DELIBERATION

Deliberation can focus on any aspect of carnivore conservation or human-carnivore relationship for which a decision or judgment is sought. At smaller scales, a local or regional government might, for example, aim to better manage compensation to citizens who have lost livestock to predators or, more generally, mitigate the unequal costs and benefits of carnivore conservation among members of the community. For such topics the scope of the community (and composition of the mini-public) can be defined judiciously and creatively. Perhaps by pairing a rural community that experiences most of the challenges of carnivore conservation

with another (non-rural) community who does not experience those challenges.

Deliberation might focus on judging the circumstances for which lethal control or hunting is allowable. Knowing the views of a representative and impartial mini-public on such issues can be just as informative as knowing the results of discourse among hyper-engaged stakeholders on the same issues.

Carnivore conservation will fail without changes at larger political and ecological scales, including new decisions by nations and groups of nations. At this scale, many issues surrounding carnivore conservation are importantly generalizable. What follows is a list of three ideas that broadly outline the scope of subjects that merit deliberation at larger geopolitical scales (in addition to smaller political and ecological scales):

1. [CARNIVORE SPECIES] living in [NAMED REGION] should be afforded *special protections* until they are *sufficiently recovered* from the threats that give rise to concern for their conservation,

where the bracketed phrases would be replaced with a specific carnivore species living in a specific region and the focus of deliberation is elucidation of the italicized phrases "special protections" and "sufficiently recovered."

2. To what extent should government proactively pursue progress toward recovery (e.g., reintroductions, habitat restoration)? Or be more limited and passive, focusing mainly on actions that merely limit further deterioration?

and

3. Should those adversely impacted by conservation be compensated (e.g., lost livestock, opportunity costs from prohibition on habitat loss)? If so, how?

Without favorable changes on these subjects, carnivore conservation will fail. Consequently, much insight would follow from knowing the views of a deliberative mini-public on these issues. These subjects are also general enough to apply to virtually all cases and readily adapted to handle the specifics of any particular case. For example, conservation of lions in Tanzania would include a strong focus on lion hunting, and conservation of clouded leopards in Malaysia would include a strong focus on habitat protection. More specific topics for deliberation can be identified in collaboration with decision makers and advocates for various positions that pertain to conservation.

With respect to idea (1), elucidating the phrase "sufficiently recovered" is meant to include (not necessarily limited to) questions about what portion of a species' historic range it should be allowed to inhabit. This consideration corresponds to a basic feature of the biodiversity crisis, which is species' loss of geographic range (Ceballos et al., 2017; Wolf and Ripple, 2017; see also *Introduction*) and has a role to play in legal-political discourse (Vucetich et al., 2006).

Acknowledging the need for deliberation on idea (1) is a remarkable acknowledgment that society has yet to develop a common response to the questions, "What is an endangered species and what do we owe endangered species?" The normative dimensions of these concerns are too great to be decided exclusively by technocrats and scientists. Additional considerations on this subject may be found in Vucetich et al. (2006), Bruskotter et al. (2014), and Vucetich and Nelson (2018).

Idea (1) sets the normative obligations and aspirations, and ideas (2) and (3) focus on the trade-offs that must be negotiated to make good on the obligations. In some cases, carnivore conservation will result in win-win outcomes, where all stakeholders are satisfied with the outcome (Redpath et al., 2013). However, there is considerable evidence that much conservation involves inescapable (and sometimes steep) trade-offs (Bowen et al., 2017; Pradhan et al., 2017; Vucetich et al., 2021), where at least one party will feel as though their interests have been unduly decided against. Where such win-lose outcomes are likely, the deliberative views of an impartial and representative mini-public are likely key for understanding the least unfair resolution.

THE PRODUCT OF A DELIBERATING MINI-PUBLIC

A mini-public may be enlisted to generate various kinds of insight, such as clarifying aspects of a difficult decision or exploring new solutions to pernicious challenges. Here, we focus on using a mini-public to better understand the views of an informed citizenry on matters of direct relevance to the policy of concern and why citizens hold those views.

These understandings are derived from responses to a survey administered to members of the mini-public before and after they deliberate. This process is also known as deliberative polling (Fishkin and Luskin, 2005). Some deliberative polling designs include polling a control group, i.e., a representative sample of those who are not part of the deliberative process. For details, see O'Malley et al. (2020).

The insights of such polling include understanding how policy-relevant attitudes are affected by knowledge of pertinent facts, normative values, deliberation, and empathy for citizens belonging to other social groups. Another important product is a report written by impartial moderators of the deliberation that summarizes the content of deliberations. This report is an additional basis (of qualitative information) for understanding the views of the mini-public.

The survey results and report are provided to decision-makers as a substantive source of insight for decision-making. As such, decision-makers and other stakeholders should be consulted about the survey design to favor eliciting the kinds of views that would be most useful to decision making.

These products of deliberation would also be provided to the general public. Doing so allows anyone to compare the attitudes of a representative group of deliberating citizens to the attitudes (or perceived attitudes) of any citizen-group within the larger community. Such comparisons can be politically significant, especially with respect to defending or criticizing subsequent decisions.

While the results of deliberation are not typically binding on decision-makers, deliberators' motivation for effortful deliberate

may depend on their belief that decision-makers will take the results seriously.

SELECTING DELIBERATORS

For heuristic value, consider an idealization where members of a mini-public are selected at random from the community. The key properties of such a mini-public would be:

- Maximization of an important principle of democracy, i.e., political equality, because every citizen has an equal chance of being selected (Fishkin and Luskin, 2005).
- If the mini-public is large enough to overcome vagarious outcomes of random sampling, then it will be demonstrably representative, within a quantifiable margin of error and with respect to key socio-demographic and ethical dispositions (Mini-publics are commonly comprised of 100 to 300 members).
- Most or all of the deliberators will not have previously engaged the subject matter to be deliberated (carnivore conservation) to any significant depth. Consequently, the identities of deliberators are not prejudicially bound to particular policy positions. As a result, members of the mini-public are more likely to possess traits essential for deliberation, i.e., greater impartiality and less partisan behavior.

In reality, perfect random selection is partially limited by (i) being unable to identify every community member to whom invitations would be randomly issued and (ii) some socio-demographic groups may be more or less likely to accept an invitation to participate.

One mitigative action for this concern is to augment the selection process with some form of quota or random sampling that is stratified according to specified socio-demography traits. Regardless of selection method, basic socio-demographic traits by which representation might be judged include: gender, race, ethnicity, religion, age, affluence, education and political orientation. For the particular case of deliberating carnivore conservation, living environment (urban, rural, suburban) may also be a salient socio-demographic trait, given that rural people are often differently affected by carnivore conservation.

To mitigate unequal rates of accepting invitations, deliberators are often offered compensation for expenses incurred by their participation (travel, lodging) and given an honorarium to at least partially compensate for the opportunity cost of participation.

Regardless of how deliberators are invited, it is important to survey members of a mini-public so that its socio-demographic composition is known. Such knowledge is a pre-requisite for taking subsequent account of any socio-demographic-based bias that may exist within a mini-public.

ENABLING DELIBERATORS' KNOWLEDGE

An essential element of deliberation is enabling deliberators to be informed of both salient facts and normative policy perspectives. An important means of doing so is to prepare a briefing booklet and make it available to deliberators prior to deliberation. Topics covered in the briefing booklet would include:

Salient Facts

This information would focus on the conservation status of the carnivore species and conflicts that arise in efforts to conserve them. Decision-makers and special interest groups should be consulted to evaluate whether all of the salient facts have been included and expressed appropriately. Uncertainties about factual claims, if present, should be communicated. Beliefs without an objective factual basis should not be presented as factual claims. Such beliefs can be presented (see below), but they should be portrayed as value-based beliefs, rather than as facts. The sometimes difficulty of distinguishing factual claims from normative views (Putnam, 2002) does not obviate the importance of making such distinctions. Failure to adequately navigate this distinction can impair the process's legitimacy (see below).

Subjects of Deliberation

The subject of deliberation should be identified as precisely as necessary to aid decision-makers. For example, deliberating about whether people should have positive attitudes about carnivores is probably too vague to be of incisive value to most real-world conservation decisions. Examples of subjects that are likely to be usefully precise include, for example, whether a particular carnivore population should be hunted, whether/how owners of depredated livestock are compensated for their losses, whether/how a landowner should be compensated if habitat protection prevents the landowner from using their land as they otherwise might have.

Appropriately Representative Range of Policy Perspectives

Policy perspectives are overtly normative. A policy perspective may align with some values and be antithetical to other values; they may favor or disfavor carnivore conservation. A policy position may be a general (e.g., making a case for the general importance of this carnivore) or specific (e.g., making a case for why and how certain stakeholders should be compensated for harms caused by carnivore conservation). The most useful expressions of a policy position go well-beyond mere assertion and include a carefully-constructed argument (e.g., Coals et al., 2019a,b; Vucetich et al., 2019).

Decisions about what policy perspectives to include should be made in close consultation with special interest groups concerned with the issue. Policy perspectives should be expressed to the satisfaction of community leaders who advocate a particular position. A critical limit on this satisfaction is that a policy position cannot be deceptive or supported by ideas presented as factual claims but lack sufficient objective factual basis.

Representation, Revisited

The composition of the mini-public is only one of two vital means through which representation is achieved. No less important to representation is the set of policy perspectives presented and details of their expression. Failure to adequately account for

this aspect of representation can impair the process's legitimacy (see below).

Limitations

Endeavoring to enable deliberators' knowledge should not be conflated with knowledge playing its hoped-for role in deliberation. One concern is that hyper-engaged stakeholders often presuppose knowledge somewhat independently of whether that knowledge is accurate. As discussed earlier in this essay, this concern tends to be mitigated when mini-publics are comprised of persons with little prior exposure to the subject being deliberated.

Furthermore, post-deliberation surveys (discussed below) should be used to assess deliberators' knowledge and it influence on their attitudes. Many published accounts of deliberative democracy report that participants tend to be adequately informed of salient facts as well as have a common understanding of salient facts (e.g., Luskin et al., 2002; Muhlberger and Weber, 2006; Grönlund and Himmelroos, 2009). Regardless, decision-makers should take account of those results when considering the views of a mini-public. Their facility in doing so would likely affect the process's legitimacy (discussed below). In any case, deliberation is fundamentally influenced by deliberators' understanding of facts, epistemic uncertainty surrounding particular factual claims, and the boundary between facts and values.

Finally, there is a perennial concern that some deliberators unduly persuade fellow deliberations, at least in part, because of their socio-economic status, communication skills, and their exercising particular ways of knowing while neglecting other ways of knowing. This concern is common to many forms of collaborative governance. For those interested to learn more about managing these concerns in the application of deliberative democracy, we recommend Beauvais (2018), Polletta and Gardner (2018), and Benson (2019).

To gain a more concrete sense for the content of briefing booklets, see Center for Deliberative Democracy (2021b).

VENUES FOR AND STRUCTURE OF DELIBERATION

Deliberation may be conducted, for example, by convening deliberators to a meeting hall for a week-end long meeting or a series of shorter meetings held over a longer period. Deliberation may also be conducted through on-line venues. On-line deliberation favors a series of shorter meeting spread over several weeks or perhaps a couple of months. While face-to-face deliberation has obvious value, so too does on-line deliberation. First, carnivore conservation is typically not afforded enough financial resources to fund face-to-face meetings. Second, the effectiveness of on-line deliberation is at least promising (Janssen and Kies, 2005; Coleman and Moss, 2012; Friess and Eilders, 2015). Third, internet-based conferencing software capable of delivering all the services required for deliberation has recently become more accessible. Fourth, the global COVID-19 pandemic has given new value to virtual meetings.

Regardless of the venue, key features of deliberation include:
Preliminaries

- Distribute briefing booklet and administer pre-deliberation survey.
- Preliminary interactions among deliberators for the primary purpose of building trust and open-mindedness.
- Succinct presentations by experts to review the background (factual) information with ample opportunity for deliberators to ask questions of experts.
- Succinct presentations by advocates for the policy positions with ample opportunity for deliberators to ask questions of the advocates.
- These presentations are intended to be reiterative of material presented in the briefing booklet and reinforce the learning of that material.

Deliberations

- Deliberation would be organized into a series of meetings with each focused on a particular topic or set of topics, such as the topics outlined in *The Subject of Deliberation*.
- The structure of each meeting might include:
  - Before meeting, remind deliberators of meeting's topic and most pertinent portions of the briefing booklet.
  - Begin meeting with a brief orientation by a meeting organizer.
  - Much of the meeting (perhaps 60 min) is for deliberation in small groups (perhaps 6–10 people) with a moderator, where ideas can be deliberated with care and an even exchange of listening and responding. The moderators' role is to make sure that deliberators stay on topic and maintain civil and inclusive discourse. These meetings are recorded. Members of small groups are selected at random from the mini-public for each meeting.
  - Some provision should be made for small groups to report the nature of their deliberations back to the entire mini-public.
  - Organizers also record new questions as they arise throughout the deliberative process. These questions would be answered as soon as possible by appropriate experts or advocates.

This outline is intended to offer a general sense for guiding participants through deliberation, not to serve as a precise planning document. For more on the implementation of a deliberative mini-public, see Grönlund et al., 2014.

Deliberations need to be led by moderators with sufficient expertise to mitigate the undue influence of unequal power dynamics among deliberators. This need is not particular to deliberative democracy; rather it is a common concern for all collaborative governance processes. As such, much has been written about the manifestations and mitigation of such power (Kadlec and Friedman, 2007; Mansbridge et al., 2010; Purdy, 2012; Choi and Robertson, 2014; Polletta and Gardner, 2018). Anyone planning to implement a deliberative process should plan accordingly.

Furthermore, in cultures with low rates of literacy and heterogeneous levels of education, much attention must be given to creating a process where existing political inequalities are mitigated, not reinforced. To this end, a fair participatory process (deliberative democracy or otherwise) requires an environment where participants can feasibly (i) become sufficiently knowledgeable of the issue and (ii) express themselves without being dominated by others. The quality of process is directly tied to these conditions. This condition turns out to be of great concern in any culture (Agarwal, 2001; Hickey and Mohan, 2004; Morales and Harris, 2014), not only cultures with low rates of literacy and heterogeneous levels of education.

Similarly, the quality of a process like that described depends on selection of appropriate subject-matter experts. Such selection should not be taken for granted. Furthermore, this concern is not particular to mini-public deliberation, but is common to many processes of collaborative governance. Little has been written about this topic; one of the few such treatments is Roberts et al. (2020). Other considerations for selection and functioning of subject-matter experts are implied by papers such as Williams (2001), Lavin et al. (2007), Rice-Bailey (2016), and Tangney (2017).

PRE- AND POST-DELIBERATION SURVEYS

Pre- and post-deliberation surveys are designed to elicit deliberators' attitudes about the subject of deliberation, as well as knowledge of salient facts and other information that may be useful in explaining heterogeneity among deliberators' attitudes. Other information might include socio-demographic traits or more basic beliefs about conservation and justice (e.g., Hülle et al., 2018; Vucetich et al., 2021).

Some survey items can be asked once (before deliberation), such as socio-demographic traits that do not change with deliberation. Some survey items can be asked twice (before and after deliberation), for cases where there is value in knowing how attitudes or knowledge of salient facts changed as a result of deliberation.

Prior research on deliberative democracy indicates that:

- Enough deliberators tend to become appropriately knowledgeable of salient facts to assess the influence of knowledge on policy-relevant attitudes (e.g., Barabas, 2004; Andersen and Hansen, 2007; Farrar et al., 2010; Esterling et al., 2011).
- Much variation in attitude change is attributable to gains in the deliberators' knowledge of salient facts (Fishkin and Luskin, 2005).
- Much of the variation in policy attitudes that remains after deliberation likely is attributable to variation in deliberators' basic beliefs (Fishkin and Luskin, 2005).
- Deliberation has illuminating effects on the tendency for some deliberators to moderate their views and others to adopt more extreme views (e.g., Wojcieszak, 2012, Lindell et al., 2017).
- Individuals who deliberate tend to display single-peaked preferences, which is a technical, but important concept in social choice theory, whereby an individual has a most preferred option and preference for alternatives decreases as the alternative is less like the preferred option (List et al., 2013, List, 2018). That condition greatly facilitates aggregation of individual preferences into a social choice that is rational. That condition should not be taken for granted.

Finally, these surveys also indicate the overall prevalence of particular attitudes for a representative sample of deliberating citizens. Prior research into deliberative democracy gives strong indication that deliberative democracy is, at least, valuable for providing a distinctively valuable understanding the human dimensions of carnivore conservation.

LEGITIMACY

Because the results of a deliberating mini-public are typically considered advisory (not binding) to decision-makers, it is important for decision-makers to take sincere and adequate account of the advice. Such accounting requires decision makers seeing the entire process as politically legitimate—from preparing the briefing booklet, to selection of deliberators, to execution of the deliberation, including the selection of subject-matter experts and advocates. An important means for evaluating political legitimacy is to ask decision-makers their views on the matter through, for example, structured interviews that account for each step of the process. For emphasis, what is to be evaluated is the legitimacy of the process without regard for the outcome of the process, before the process is executed.

Assessing political legitimacy from the perspective of the general public is also valuable. If, for example, decision-makers neglect the results of deliberation, but the general public indicates that the process is politically legitimate, then members of the general public may use the results of deliberation to pressure decision-makers or appeal to the general public.

Special interests' views on the political legitimacy of the process is also likely important to assess. For example, an influential special interest can obstruct decision-makers in applying the results of deliberation. If, however, a special interest acknowledged the political legitimacy of the deliberative process in advance of knowing its result; then the special interest would, at least, have less public justification for being obstructionist.

Successful deliberation requires deliberators to be motivated to do the demanding work of deliberation. That motivation is favored by deliberators' belief that the results of their work will be taken seriously. That belief can be fostered by acknowledgments of the process's political legitimacy.

Finally, political legitimacy is a broad and complex topic for political science in general, as well as more specific domains such as environmental governance and deliberative democracy. Readers unfamiliar with this literature can find access to it through papers such as Buchanan (2002), Parkinson (2003), Hogl et al. (2012), and Fabienne (2017).

WOULD DELIBERATIVE DEMOCRACY FAVOR CARNIVORE CONSERVATION?

Deliberative democracy is distinctive among various modes of collaborative environmental governance. First, deliberative

democracy is better-designed than other processes to handle cases where the object of conservation (carnivores) is not sufficiently valued by those who are most detrimental to its conservation. Carnivore conservation is, of course, emblematic of such cases. Second, deliberative democracy engenders a rich kind of representation and impartiality that is far more difficult to achieve through processes that focus on managing conflicts among hyper-engaged stakeholders. Deliberative democracy also has a track record of being successfully applied to complicated issues across a wide range of cultural settings, including within deeply divided societies (e.g., Luskin et al., 2014).

While deliberative democracy is appreciated for its positive features—most generally its claims to being representative and deliberative—it does draw criticism. Two criticisms of general importance and relevant to conserving biodiversity are (Dryzek, 2000; Young, 2001): First, the conditions for genuine representation and deliberation are too difficult to reliably achieve in most real-world settings. Second, while an adequately implemented mini-public will claim to favor—as a matter of principle—a *procedurally just* outcome; there is no assurance that it will result in a *just outcome* as perceived by any particular group (Vucetich et al., 2018). For additional limitations on the role of deliberation in the environmental space, see Flynn (2009).

Returning to the specifics of carnivore conservation, two questions merit attention:

1. Under what conditions, if any, is a large diverse community represented by a deliberative mini-public likely to favor carnivore conservation, even when doing so is against the expressed interest of some members of the larger community?
2. If the views of a deliberative mini-public are not binding, what influences might prevent those views from being manifest?

The second question is likely easier to answer than the first. The views of a mini-public may be disregarded if the process's legitimacy was not established and (or) if a special interest uses its power to obstruct the decision-making process. The undue influence of special interests (and corruption) in conservation politics extends far beyond concerns particular to deliberative democracy. In the United States, for example, there is strong support for the Endangered Species Act (ESA), regardless of political orientation (Bruskotter et al., 2018b; Offer-Westort et al., 2020). Yet, special interests have maintained consistent pressure on Republican lawmakers to dismantle the Endangered Species Act (ESA) and Democratic presidential administrations to weaken the ESA (Vucetich and Nelson, 2014; Center for Biological Diversity, 2015; Bruskotter et al., 2018a).

With respect to the first question, it is important to acknowledge that most conservation challenges are multifaceted, requiring multiple approaches. As such, we do not suppose that simply inserting a mini-public into a governance process would be sufficient by itself to solve hardly any conservation problem. Nor do we suppose that a mini-public is the critical missing tool for every conservation problem. Rather we suppose that it may be especially useful for cases, and where (i) those who most harm a species are politically overrepresented, as discussed in One *Concern*, and (ii) hyper-engaged stakeholders obstruct sought-after levels representation and impartiality, as discussed in *A Second Concern*. Where conservation cannot be advanced without, for example, better enabling a government to more effectively manifest citizens' will, then employment of a mini-public by itself would be insufficient.

While those concerns are common to many conservation problems, they are not the limiting obstacle to all conservation problems. For example, some cases are limited by governments that are willing to enact conservation supported by democratic processes, but unable to do so due to limited power (e.g., as in the presence of certain instances of transnational crime). We expect mini-public to be useful to the extent that a particular conservation case is limited by the two above-mentioned concerns.

That first question—about conditions under which deliberative democracy is likely to be successful—raised other considerations and difficult questions. In particular, existing research suggests that representative samples of citizens tend to be supportive of carnivore conservation (Williams et al., 2002; Bruskotter et al., 2018b) and sensitive to the needs of those whose physical and financial well-being is genuinely impacted by conservation (Vaske et al., 2013; Slagle et al., 2017). If those judgments are even approximately accurate, then it also seems plausible that the views of a mini-public will favor carnivore conservation when the details of the case represent a win-win scenario (*sensu*, Redpath et al., 2013).

Far less certain, it seems, would be cases that involve win-lose outcomes, where at least one party within the larger community believes that certain outcomes would leave their interests (or carnivores' interests) unduly overridden. One concern with such cases is the difficulty of reliably distinguishing win-win and win-lose scenarios, and distinguishing outcomes that involve a "loss" from those that are unfair.

Furthermore, views on fair and just adjudication of conflicts between humans and nature vary widely among writers (e.g., Vucetich et al., 2018; Washington et al., 2018; Treves et al., 2019), but little is known about the prevalence of such views among the planet's citizens. While much is known about the social psychology of social justice—which is concerned with adjudicating conflicts that include only human interests (e.g., Miller, 1999), very little is known about the social psychology pertaining to the fair and just adjudication of conflicts between humans and nature (but see Vucetich et al., 2021).

These connections to justice are useful for another reason. Specifically, justice may be evaluated according to the procedure that led to an outcome and (or) the outcome itself. If the deliberations of a mini-public are properly executed, then there is an arguable (not indisputable) sense by which the outcome is procedurally just. The question, without a broadly agreed upon answer, is, "What does outcome justice look like when adjudicating the myriad ways for which the interests of humans and non-human nature conflict?"

The general failure of carnivore conservation allows for the possibility that deliberative democracy, which has not to our knowledge ever been applied to carnivore conservation, is essential (if not sufficient) for realizing procedural justice and outcome justice—a condition that might be called the flourishing coexistence between carnivores and humans. And,

even if deliberative democracy failed, the nature of the failure would be richly insightful. In any case, the broad failure of current carnivore conservation cries for the need to try something different. To that end, deliberative mini-publics stands out.

DATA AVAILABILITY STATEMENT

The original contributions generated for the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

JV led the development of the manuscript. DM and JB contributed to conceptual refinement and writing. All authors contributed to the article and approved the submitted version.

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Using a Conflict Framework Analysis to Help Beekeepers and Giant Armadillos (*Priodontes maximus*) Coexist

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Damage caused by wildlife is one reason preventing peaceful coexistence between humans and wildlife. To identify the complexity and scope of human-wildlife interactions and to guide conservation interventions, a theoretical framework has been recently proposed, based on the field of conflict analysis and peacebuilding. Despite its importance, to our knowledge, there are no studies yet testing the framework. We therefore adapted and expanded the framework to investigate a wildlife-people interaction scenario, involving damage by giant armadillos (*Priodontes maximus*) to apiaries in the Brazilian Cerrado biome of Mato Grosso do Sul state. From August to November 2018, we interviewed 111 beekeepers identified through a mixed random and snowball sampling design to assess at which level of conflict this interaction could be framed, and to identify strategies able to promote coexistence. Analysis of the five key areas of the framework suggested the current human-wildlife interaction is a level one conflict. This means the negative relation between beekeepers and giant armadillos is still not rooted in less visible, more complex social disagreements, but founded in a material dispute: destruction of beehives. We used the findings to create an intervention strategy which involves: (i) the implementation of mitigation strategies that prevent giant armadillos from predating beehives; (ii) a certification scheme to acknowledge beekeepers' efforts to implement the mitigation strategies, and (iii) a Citizen Science Program using an app that enables data gathering for adaptive management, as well as maintains beekeeper engagement. We hope beekeepers-giant armadillos' coexistence will become beneficial rather than a challenge with the novel interventions.

Keywords: beekeeping, coexistence, conflict analysis, human-wildlife conflicts, wildlife damage management

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INTRODUCTION

Damage caused by wildlife is one of the reasons preventing the peaceful coexistence between humans and wildlife (Kansky et al., 2014), because it may lead to material and economic losses, threaten livelihoods (Messmer, 2009), and even impact the mental health of the people involved (Thondhlana et al., 2020). Retaliatory killing of animals blamed for damage may follow (Cerri et al., 2017), which translates into a worldwide threat to numerous species (Woodroffe et al., 2005).

Due to that, conservationists and wildlife managers have searched means to manage such challenges. One approach is to directly tackle negative impacts caused by wildlife, with two main strategies commonly adopted in those situations. The first is damage mitigation which seeks either to influence the behavior of the damaging animal (e.g., deterrents for crop-raiding elephants; King et al., 2011), or to reduce the vulnerability of the target (e.g., preventive husbandry; Dickman et al., 2018). The second approach relies on economic instruments to offset impacts, such as insurance (Chen et al., 2013) and financial compensation for the losses incurred (Bauer et al., 2017).

Yet, measures to reduce damage or to provide financial benefits may not necessarily lead to the desired or hoped for peaceful coexistence (Naughton-Treves et al., 2003). Previous evidence indicates the levels of wildlife damage or losses experienced are not always directly related to retaliation or to resentment levels (Dickman, 2010; Marchini and Macdonald, 2012; Kansky et al., 2016). Other factors may impair coexistence, including non-tangible characteristics of a person (e.g., empathy for the species), society (e.g., social identity), culture or institution (see Pooley et al., 2017; Thondhlana et al., 2020). For example, Cavalcanti et al. (2010) found that human persecution to jaguars in Brazil was better explained by a cultural status associated with jaguar hunting than by the economic impacts of their livestock depredation. Neglecting these less visible and non-tangible elements may constrain conflict identification and hinder appropriate solutions, therefore leading to ineffective interventions (Dayer et al., 2019), or even raising negative perceptions of the species leading to escalating human retaliatory practices (Madden, 2004).

To overcome the limitations of current conservation approaches to conflicts, at least two studies have argued for the importance of incorporating the principles and processes of both conflict analysis and peacebuilding research fields. Madden and Mcquinn (2014) claim the knowledge from these fields may provide tools to qualify the challenges of human-wildlife conflicts and coexistence, as well as can guide more effective interventions. The authors propose an analytical model that classifies three levels of conflict. The first is when material and financial losses are at the core of the problem; therefore, preventing or compensating these losses suffices to alleviate such conflicts. The second level is when the people affected by damage may hold strong resentment toward the responsible species, or to other stakeholders addressing the issue (e.g., conservationists). Interventions at this level are less straightforward and may require strengthening relationships between stakeholders involved. Finally, the third level is when a deep-rooted conflict is at play, involving antagonistic values, beliefs or identities. In this case, interventions require complex approaches to reconcile divergent perceptions.

Based on Madden and Mcquinn (2014), Zimmerman et al. (2020) advance on practical guidance on how to identify the conflict level of a given negative interaction between humans and wildlife. The authors suggest investigating five key areas: (i) perception of the species blamed; (ii) exploration of previous attempts to solve the situation; (iii) questions about the situation itself; (iv) the extent of people's willingness to find solutions, and (v) views about others involved in the context or trying to assist with solutions.

Despite the proposed framework relevance to human-wildlife interactions, to our knowledge, the structure of investigation described by Zimmerman et al. (2020) has not been ground-tested in a problematic human-wildlife coexistence situation. Therefore, in this article, we advance from this study and the current conservation literature in two ways. First, we extend the proposed framework by suggesting how to assess the five key areas of investigation using qualitative and quantitative social research methods. Second, we apply the framework to investigate whether damage by giant armadillos (*Priodontes maximus*) to apiaries in the Brazilian Cerrado biome can be characterized as conflict and, if so, at what level. Understanding interactions between beekeepers and giant armadillos may allow practitioners in the conservation field to design more effective interventions to help promote a peaceful coexistence between humans and wildlife.

METHODS

The Human-Giant Armadillo Interactions and Study Site

The study was conducted in the giant armadillo's distribution within the Cerrado biome of Mato Grosso do Sul (MS) state of Brazil (**Figure 1**) (Ferraz et al., 2021), where honey production is an important activity. The state has the highest production per hive/year in Brazil, 34 kg above the 18 kg national average (FEAMS – Federação de Apicultura e Meliponicultura do Mato Grosso do Sul, 2017). Beekeeping has also been steadily increasing in family agricultural practices worldwide, due to its potential inclusiveness amongst smallholders (Gonçalves, 2006).

In the Cerrado biome of MS, beekeepers place hives along the edges of native vegetation remnants to allow wildflowers' visitation by bees. Brazil's Cerrado is a highly diverse savanna ecosystem, which provides critical habitat for several endemic and rare species (Klink and Machado, 2005). Yet more than half of the original Brazilian Cerrado has been transformed into pasture or cash crop agriculture, whereas only 19.8% remains undisturbed (Green et al., 2019). In MS, there are 58,459 km^2 of remaining Cerrado left, which is 16% of the total state area (Reynolds et al., 2016). The remaining MS Cerrado is highly fragmented and predominantly found in small patches (Reynolds et al., 2016), where giant armadillos still survive (Ferraz et al., 2021).

Giant armadillos are the largest living species of their kind, with adults weighing up to 60 kg and measuring up to 1.5 m long (Carter et al., 2016; Desbiez et al., 2019). Despite their large size, giant armadillos go frequently unnoticed due to their solitary, nocturnal, and fossorial habitats (Eisenberg and Redford, 1992; Desbiez et al., 2020a). The species feed mainly on ants and termites (Anacleto and Marinho-Filho, 2001), but may opportunistically consume bee larvae (De Melo and Nogueira, 2020). Giant armadillos have learned how to knock over beehives in apiaries to feed (Desbiez et al., 2020b), often imposing substantial economic losses to beekeepers. To overcome this

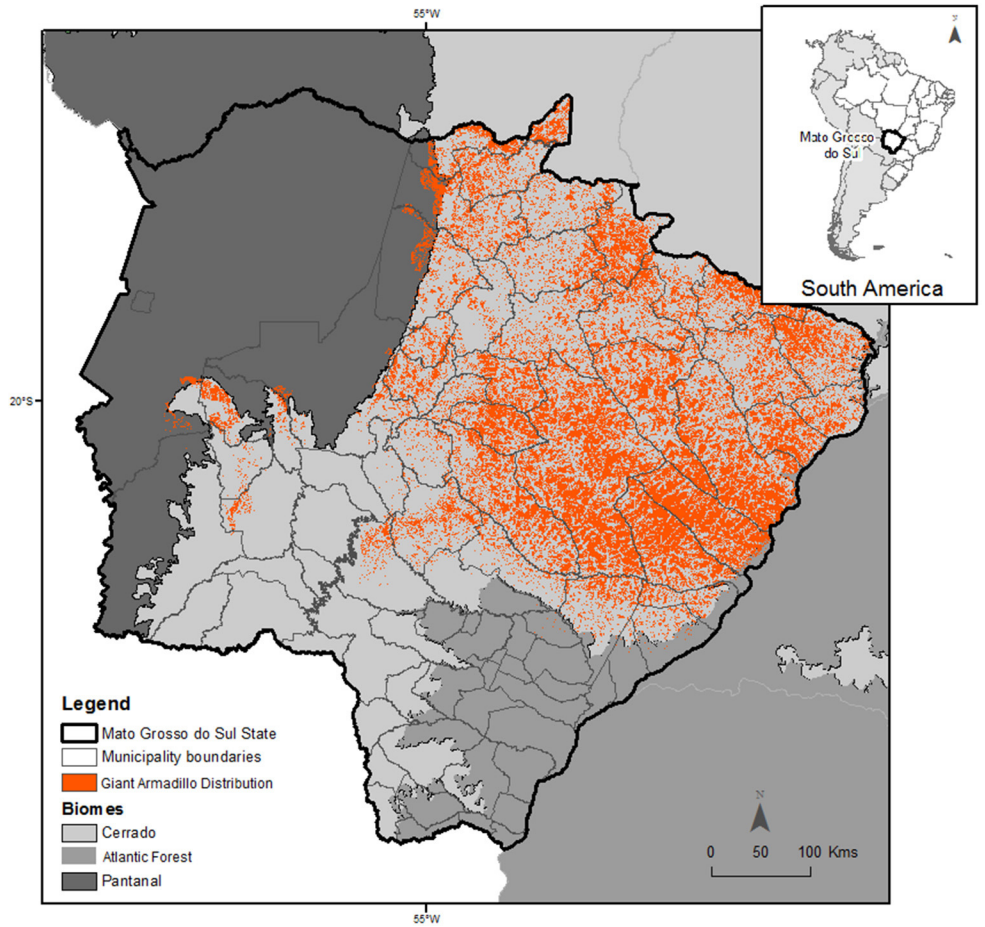


FIGURE 1 | Giant armadillo (*Priodontes maximus*) distribution in the Cerrado biome of Mato Grosso do Sul state, Brazil (Ferraz et al., 2021).

problem, some beekeepers implement non-lethal strategies such as raising hives or electric fences (for details, see Desbiez et al., 2020b) while others use lethal methods, such as poison. Indeed, giant armadillos are highly susceptible to poisoning because, after destroying beehives, they return to feed on them for several nights. Given giant armadillos' low population densities and low population growth rates (Desbiez et al., 2020a), besides their national/global "Vulnerable" IUCN classification (Anacleto et al., 2014), the loss of a single animal to poisoning can precipitate local extinctions (Desbiez et al., 2020b).

Data Gathering

To investigate the human-wildlife interaction, our first step was to identify the beekeepers living in giant armadillos' area of occurrence within the Cerrado of Mato Grosso do Sul state. Due to the large dimension of this area, we relied on registration lists of honey producers kept by their association. Using these lists, we randomly selected one beekeeper per municipality ($n = 19$) to be interviewed. However, not all beekeepers were enrolled in the associations, even when they employed equivalent honey management practices. Therefore, this procedure could bias our sample. To avoid biases we combined the random sample from these lists with identification through non-random snowball sampling (Goodman, 1961). Snowballing is an appropriate method when dealing with subjects difficult to access (Newing, 2011), as was the case with non-associated beekeepers. To do that, associated interviewees were asked to name another beekeeper, who then provided another name and so on.

Using a semi-structured interview guide, which was previously pilot- tested (April 2018), we carried out face-to-face interviews with 111 beekeepers from August to November 2018. Information gathered included: sociodemographic information (sex, age, schooling, origin), income sources, beekeeping characteristics (e.g., involvement period; associated or not), and aspects related to the coexistence with giant armadillos. To characterize and better understand the challenges beekeepers face by producing honey alongside giant armadillos, we followed the guidelines in the Zimmerman et al. (2020) framework, thus incorporating the five key areas of analyses. Because the framework did not specify how key areas should be assessed, we devised indicators and associated measures for each of them using quantitative and qualitative social research

TABLE 1 | Indicators devised to assess each of the five key areas of conflict analysis proposed by Zimmerman et al. (2020), data gathering and analyses.

Key areas of conflict analysis	Indicators used	Data gathering	Data analyses
1. Perception of the species	i. Beekeepers' views about giant armadillos	Free listing of views about giant armadillos (3 words)	Smith's salience (combination of order and frequency of citation) Word cloud analysis
	ii. Perceived benefit of giant armadillo's existence	Open question: Are there any benefits about giant armadillo's existence? Probes for positive answers: What is (are) such benefit (s)?	Inductive content analysis and frequency of response's categories
	iii. Attitudes toward giant armadillos	Likert scale composed of six statements Five points response scale (i.e., from totally agree to totally disagree)	Scale reliability (Cronbach alpha)
	iv. Not in my back yard sentiment	Closed question: Which destination do you prefer for the giant armadillo in the future? (five response categories)	Frequency of response categories
	v. Understanding for the needs of giant armadillos	Open question: Why do you think giant armadillos destroy the hives?	Inductive content analysis and frequency of response categories
2. Conflict situation itself	i. Limitations to beekeeping	Open question: What are the main current and past (5 year ago) limitations to beekeeping?	Inductive content analysis and frequency of response categories
	ii. The severity of giant armadillo damage to beehives	Absolute number (last 12 months and 5 years ago) Percentage of the total production (last 12 months)	Descriptive analyses
	iii. Attitudes toward giant armadillo persecution	Likert scale composed of six statements (using target, action, context and time- specific). Five points response scale (i.e., from totally agree to totally disagree)	Scale reliability (Cronbach alpha)
	iv. Relationship between damage and attitudes toward giant armadillos	Damage: number of beehives destroyed in the last 12 months Attitudes: Likert scale composed of six statements	Kruskal-Wallis test *plus* Bonferroni *post-hoc* Multinomial logistic regression
	v. Relationship between damage and attitudes toward giant armadillos' persecution	Damage: number of beehives destroyed in the last 12 months Attitudes: Likert scale composed of six statements	Kruskal-Wallis test *plus* Bonferroni *post-hoc* Multinomial logistic regression
	vi. Beekeepers' tolerance to damage by giant armadillos	The proportion of individuals who have positive attitudes toward a species despite suffering damages from the same species	Tolerance to damage index
3. History of attempts to solve the conflict	i. Beekeepers' strategies to prevent damage by giant armadillos	Number of beekeepers using non- lethal and lethal strategies Strategies ranked in low, medium, and high effectivity	Coding and frequency of response categories
4. Willingness to find solutions	i. Beekeepers' willingness to adapt management practices	Closed question: How willing would you be to adapt your management practices to stop losing hives to giant armadillos?	Frequency of response categories
5. Others involved in the issue	i. Interest in receiving assistance by third parties ii. Perceived image of third parties iii. Other groups'perceptions	Closed question: Would you like to receive any assistance to alleviate the challenge of working alongside giant armadillos? Probes: Can you tell me more about that? and Who do you think this help would come from?	Frequency of response categories Inductive content analysis

methods, as described further (see **Table 1** for a synthesis of data gathering and **Supplementary Material 1** for data analyses details).

Extending the Framework: Devising Indicators

The number of indicators and the extent to which we investigated subjects in each key area reflected the results of our interview pilot-test. When questions proved useless to elicit important information, we dropped them to shorten our interview.

- First key area: beekeepers' perceptions of giant armadillos

Perception is a concept that can mean different things depending on the discipline. We, therefore, adopted five perception indicators to cover the range of conceptualizations in the Human Dimensions' literature. First, beekeepers' views about giant armadillos were investigated through a free listing of the first three words that came to the interviewee's mind when thinking about giant armadillos. The procedure relies on the rationale that words are symbolic representations of concepts, places or

objects (Carlston, 2013), allowing inferences about the cultural salience of the items in a given domain. Second, the perceived benefit of the species' existence, an aspect recognized as crucial for enhancing wildlife conservation (Bennett et al., 2017). Third, beekeeper's attitudes toward giant armadillos, and therefore their tendency to respond with some degree of favorableness (or not) to the species (Eagly and Chaiken, 1993). In Social Psychology, attitudes are considered important to explain how people think, feel and respond to wildlife damage (Decker et al., 2012). Fourth, the so-called *Not in My Back Yard* (NIMBY) sentiment, which considers people's opposition to the existence of undesirable things/species in their own neighborhood (Dear, 1992). Finally, fifth, beekeepers' understanding of giant armadillos' needs, as a measure of the level of comprehension of the species' needs.

- Second key area: the conflict situation

This key area requires understanding whether people's complaints arise from tangible impacts and experienced losses. To assess this, we first investigated beekeepers' perceptions of their current limitations to beekeeping. Because respondents tend to interpret a given question in light of previous ones, to avoid biases, we posed this question at the interview beginning and before talking about damage by giant armadillos. Second, we asked about the severity of giant armadillo damage to beehives, to assess the intensity of the problem. Third, we assessed attitudes—i.e., how favorable a person was—toward giant armadillo persecution. This information provides insights on how beekeepers would behave in response to giant armadillos' damages (following the Theory of Planned Behavior (Ajzen, 1985). Fourth, the relationship between damage and attitudes toward giant armadillos and their persecution was questioned, to verify if the damage was an important predictor of attitudes. Fifth, we assessed beekeepers' tolerance to giant armadillo's damage. Tolerance is often associated with attitudes in wildlife management literature to understand HWC (Frank, 2016).

- Third key area: previous attempts to address the conflict

We assessed both beekeepers' lethal and non-lethal strategies to prevent giant armadillos' damages, and also gathered information on their effectiveness.

- Fourth key area: willingness to adapt to alleviate conflict

We used a single-item measure to obtain a direct evaluation of beekeepers' openness to change management practices to reduce conflict.

- Fifth key area: other people involved in the issue

To investigate beekeepers' perception of other human groups involved, we assessed their openness to receive assistance from third parties (from I *would like very much* to *I wouldn't like*). Because trust in a person or group delivering an intervention predicts people's levels of cooperation (Baynham-Herd et al., 2020), we also gathered information on whom they believe this help should and would or not came from.

To investigate perceptions of other human groups involved (e.g., conflictive or not), we first identified the stakeholders in the beekeeper's narrative, including: (1) government authorities responsible for sanitary and safety inspections of hives; (2) eucalyptus plantation companies and landowners, which allow beekeepers to use their lands; (3) federation and associations of beekeepers, and (4) other wildlife conservation groups. We then conducted unstructured interviews (Newing, 2011) from January 2019 to January 2020 with these groups. The interviews were conversations arranged in advance to talk about the role, knowledge, understanding, and consequences of the negative interactions between giant armadillos and beekeepers.

The most representative government authority in this context is CSEAP-MS- Câmara Setorial Consultiva da Apicultura de Mato Grosso do Sul. A.L.J.D. participated in five meetings with representatives of this agency. The perception of the two silvicultural companies operating in the region was obtained through 10 virtual meetings over a year with responsible employees. Ten beekeeping associations situated across the giant armadillo's range in MS Cerrado were consulted through phone interviews with their representatives. Regarding NGOs activities, although the category was superficially mentioned in a few beekeepers' interviews, no organization was named, and we did not find further evidence about any NGO operating on beekeeping issues in the region.

RESULTS

Applying the Framework

Beekeepers' Characteristics

Among the 111 interviewees, less than half (43.24%) were born in the study area, and almost all were male (99.1%) with an average age of 50.1 years old ($\pm$12.54; range = 27–89). This gender bias reflects the overall gender division of labor in the region for beekeeping. Most women do not participate in beehives' management, although they may be involved in honey-selling and bottling. Interviewed beekeepers had, on average, 7.4 $\pm$ 3.2 years of schooling, although about a third of them (32.4%) studied <4 years. Beekeeping was the main source of family income for 41.1% of the interviewees, who relied on honey for half or more of their income. Most beekeepers (64%) had over 10 years of experience in this activity, whereas only 7.2% had <5 years. Most beekeepers (60%) were members of beekeeping associations.

Beekeepers' Perceptions Toward Giant Armadillos

Beekeepers' views about giant armadillos tended to be neutral, as evidenced by the combination of the frequency and rank order in the free listing (Smith's salience). Neutral aspects (e.g., "big," "strong," "rare," "high") were the items most frequently reported—around three times more than either positive (e.g., "beautiful") or negative (e.g., ugly) aspects (**Table 2**).

On average, beekeepers had favorable attitudes toward giant armadillos (**Table 3**), and most beekeepers (82%) regarded giant armadillos as beneficial, albeit about half of them could not specify why (often they provided vague answers: "must have, every species has"). Among the benefits identified, pest control (i.e., eating social insects) was the most frequently mentioned (16%), followed by benefits associated with burrowing (i.e., seed

germination and dispersal, ecosystem engineering, 11%), and species' intrinsic value (i.e., value irrespective of use/service, 9%).

The analysis of the "not in my backyard" sentiment (NIMBY; **Figure 2**) indicated most beekeepers do not believe giant armadillos should be eradicated either globally or locally, though they did wish fewer animals occurred in their neighborhoods. In contrast, some of them stated they would prefer that giant armadillos' population remained stable or even increased in their region.

Finally, when asked about the reasons why giant armadillos destroy hives, 97.3% of beekeepers replied that the animal does so because it needs feeding. In contrast, very few beekeepers (2.7%) believed giant armadillos typically have a destructive-driven behavior.

TABLE 2 | Frequency, average rank, and salience of beekeepers' perception about giant armadillos (*Priodontes maximus*) through free-lists.

Category	Frequency (%)	Average rank	Salience
Neutral	94.2	2.54	0.447
Positive	39.8	1.71	0.298
Negative	30.1	2.03	0.193

TABLE 3 | Descriptive statistics of beekeepers' attitudes toward giant armadillos (*Priodontes maximus*) and toward its persecution.

Variable	Mean ± SD	Median	Min[a]	Max[b]
Attitudes toward giant armadillos	3.51 ± 6.1	4	−11 (−12)	12 (12)
Attitudes toward giant armadillos' persecution[c]	−5 ± 4.4	−8	−12 (−12)	8 (12)

[a] *Minimum value observed and minimum possible value observed (in parenthesis).*
[b] *Maximum value observed and maximum possible value observed (in parenthesis).*
[c] *Negative values indicate unfavorability to persecution; positive values indicate favorability to persecution.*

Conflict Situation

Over a quarter (27.9%) of the interviewees cited damage by giant armadillos as a current limitation to beekeeping, second only to pesticide use cited by 39.6%. When enquired about the main limitation to beekeeping in the prior 5 years, almost half (48.6%) of the respondents stated giant armadillos were their biggest problem then. Despite that, 46% per cent of respondents reported at least one beehive damage by giant armadillos in the previous 5 years, equivalent to the rate (44.14%) reported for the prior 12 months. Thirty-eight percent of interviewees who had experienced damage reported losing 25–50% of their beehives in the last 12 months, whereas 42% lost <25% and 20% lost >50% of their beehives.

The damage was an important factor explaining beekeepers' attitudes toward giant armadillos. When comparing attitudes toward the species among three groups of beekeepers (no damage, low damage and medium/high damage), results from a Kruskal-Wallis test indicated a difference ($X^2 = 11.981$, d.f. $= 2$, $p < 0.001$) between the mean ranks of at least one pair of groups. Comparison between three pairs of groups evidenced a consistent difference on attitudes between beekeepers who experienced medium to high damage. Dunn's pairwise tests for the three pairs of groups evidenced a consistent difference ($p < 0.001$) on attitudes between beekeepers who experienced medium to high damage (H = 35.53) and those that did not (H = 64.36). However, there was no evidence of difference between the other pairs ($p > 0.05$).

The Kruskal-Wallis test also indicated there was an effect of the group on attitudes toward the giant armadillo persecution [$X^2 = 14.997$; d.f. $= 2$, $p < 0.001$]. Dunn's pairwise test showed that beekeepers groups differed in how much they agreed with the persecution of giant armadillos ($p < 0.001$). Beekeepers who experienced medium/high damage (H = 81.56) differed from those who did not (H = 48.67). The same result was found when comparing the group that experienced low damage (H = 57.64) and the one that experienced medium to high damage. There was

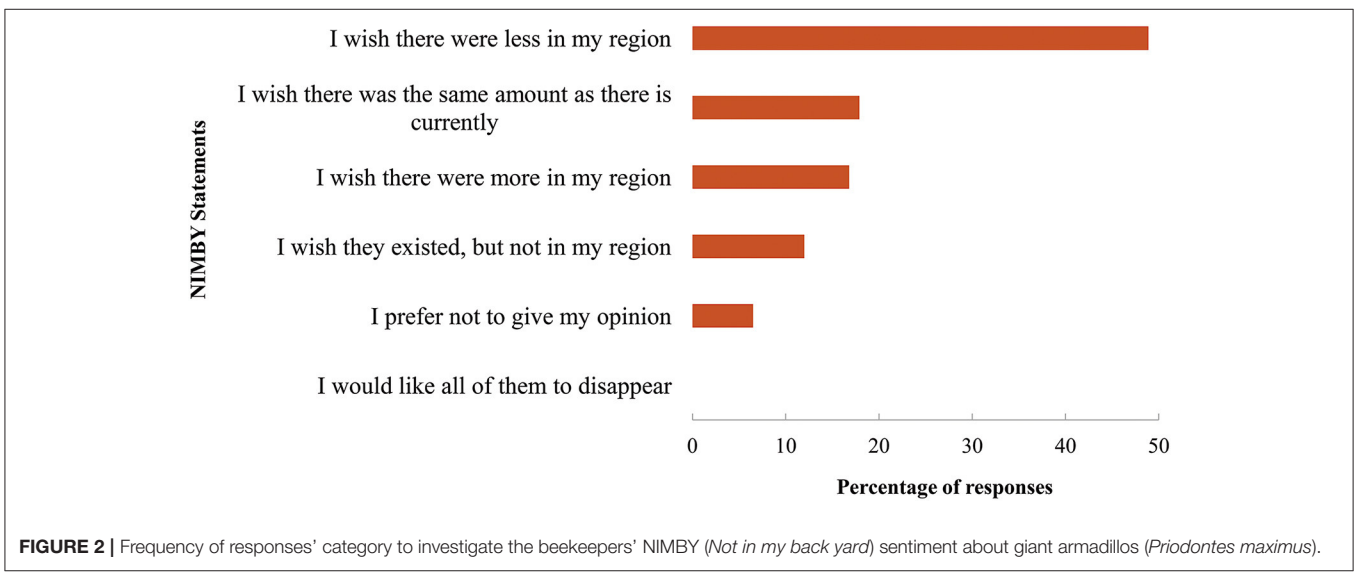


FIGURE 2 | Frequency of responses' category to investigate the beekeepers' NIMBY (*Not in my back yard*) sentiment about giant armadillos (*Priodontes maximus*).

no evidence that beekeepers' attitudes differed between those that experienced low damage and those that did not ($p > 0.05$).

The results of the association between damage and attitudes toward giant armadillos (multinomial regressions) display a similar trend. The first multinomial regression tested the importance of damage on beekeepers' attitudes toward giant armadillos (**Table 4**). Pearson [$X^2_{(208)}$199.287, $p = 0.656$] and Deviance chi-square [$X^2_{(208)}$157.454, $p = 0.996$] tests indicated that this regression model fits the data well and supported the existence of a relationship between the damage extent in the prior 12 months and attitudes toward giant armadillos [Model X^2(d.f. = 12) = 44.396, $p < 0.001$; classification accuracy rate 69.4% higher than the proportional by chance accuracy rate]. Damage was a consistent predictor of attitudes toward the species as beekeepers scoring higher on this variable were more likely to hold negative attitudes toward the species (b = 0.54, S.E. = 0.016, $p < 0.001$, odds = 1.055). The odds ratio indicates that an increase in damage of 1 beehive destroyed would increase the chance that the beekeeper holds negative attitudes toward giant armadillos by a factor of 1.055. Beekeeping experience (b = 0.051, S.E. = 0.026, $p < 0.05$, odds = 1.052) and age (b = 0.49, S.E. = 0.024, $p < 0.05$, odds = 1.050) were also significant predictors in the model. Years of formal education, being a member of the beekeeping association and having beekeeping as a main source of income did not correlate with attitudes toward the species.

According to the goodness-of-fit tests (Pearson [$X^2_{(206)}$168.337, $p = 0.974$] and Deviance chi-square [$X^2_{(206)}$105.679, $p = 0.999$] of the second multinomial regression model (**Table 4**), data supported the effect of damage on attitudes toward giant armadillo persecution [Model X^2(d.f. = 12) = 45.882, $p < 0.001$; classification accuracy rate 71.2% higher than the proportional by chance accuracy rate]. Extension of damage and social norms -i.e., one's perception about what constitutes appropriate conduct by the own peers- contributed significantly to the model. In contrast, beekeepers' age, beekeeping as the main source of income and beekeeping experience did not contribute to the model. Beekeepers who experienced damage were 1.048 times more likely to be in the favorable to persecution group than in the unfavorable group. Moreover, believing that other beekeepers are favorable and would approve giant armadillos' persecution (i.e., social norms) increased the odds of being in the favorable to persecution group by 5.23.

Non-negative attitudes toward the species were presented by 65.3% of beekeepers who experienced damage, showing a Tolerance to Damage Index of 0.09, indicating high tolerance to giant armadillo damage.

TABLE 4 | Multinomial logistic regressions of the association between beekeepers' characteristics/experience and: (i) attitudes toward giant armadillos (*Priodontes maximus*) (Model 1) and (ii) attitudes toward their persecution (Model 2).

				95% Confidence Interval for Exp(B)	
Response variable group	**Predictor variable Exp**	***(B)***	***P***	**Lower Limit**	**Upper Limit**
Model 1[a]					
Negative attitudes	Intercept		0.028		
	Damage	1.055	0.001*	1.024	1.088
	Schooling years	0.841	0.084	0.692	1.023
	Beekeeping experiene	1.052	0.047*	1.001	1.107
	Being part of an association	0.887	0.839	0.279	2.820
	Age	1.050	0.044*	1.001	1.101
	Beekeeping as the main source of income	0.573	0.623	0.062	5.274
Neutral attitudes	Intercept		0.252		
	Damage	0.971	0.470	0.898	1.051
	Schooling years	0.943	0.550	0.778	1.143
	Beekeeping experience	1.096	0.001*	1.039	1.156
	Being part of an association	1.049	0.944	0.275	3.996
	Age	0.989	0.671	0.939	1.042
	Beekeeping as the main source of income	2.98	0.273	2,98E-06	2,98E-06
Model 2[b]					
Favorable to persecution	Intercept		0.052		
	Damage	1.048	0.035*	1.003	1.096
	Schooling years	0.987	0.933	0.725	1.343
	Beekeeping experience	1.100	0.033*	1.008	1.201
	Age	1.054	0.270	0.960	1.159
	Beekeeping as the main source of income	0.121	0.062	0.013	1.113
	Social norms	5.232	0.003*	1.780	15.380
Neutral to persecution	Intercept		0.028		
	Damage	1.046	0.007*	1.012	1.081
	Schooling years	1.047	0.649	0.860	1.274
	Beekeeping experience	0.993	0.810	0.935	1.054
	Age	1.040	0.136	0.988	1.095
	Beekeeping as the main source of income	0.404	0.174	0.109	1.493
	Social norms	1.274	0.402	0.723	2.244

[a] *Reference category: positive attitudes.*
[b] *Reference category: unfavorable to persecution.*
**P < 0.05.*
Pseudo R^2 (Model 1) = 0.39.
Pseudo R^2 (Model 2) = 0.46.
Log likelihood (Model 1) = 157.454.
Log likelihood (Model 2) = 105.679.

History of Attempts to Prevent Giant Armadillo Predation

While some beekeepers adopted non-lethal mitigation strategies, others used lethal methods to exterminate hive-damaging animals, including poisoning and trapping. Beekeepers reported that trapping was almost always unsuccessful, time-consuming, and hence frustrating. Poisoning, instead, was considered very successful, as giant armadillos returned to feed on the fallen bee combs in the following nights after an attack.

Almost all (94.6%) beekeepers implemented voluntarily, in the previous 12 months, at least one among 10 non-lethal methods intended to prevent giant armadillo attacks. Three of them were considered highly effective, despite only 30.6% of beekeepers

have attempted to implement them, while four were considered moderately effective and three little effective. Less than half of the interviewees (41.4%) previously tested at least one among the low effective measures which failed to prevent giant armadillo damages to their beehives.

Willingness to Find Solutions

Almost all beekeepers (96.4%) were willing or very willing to make changes in how they raise bees so as not to lose more hives to giant armadillos.

Other Parties Involved

The large majority (82%) of interviewed beekeepers would like or would like very much to receive third-party assistance to end giant armadillos' predation on hives. We identified four potential stakeholders in beekeepers' narrative: government agencies, beekeeper's associations, silvicultural companies, and NGOs. None of them was negatively viewed by beekeepers, although they were somehow ambivalent about the perceived trustworthiness in receiving support from government agencies. While certain government local agencies instilled more confidence (e.g., SENAR—Serviço Nacional de Aprendizagem Rural and AGRAER—Agência de Desenvolvimento Agrário e Extensão Rural), others were considered suspect. Even so, beekeepers did not deny future collaboration with them to devise conflict solutions. Beekeeping associations, where they exist, are active and respected institutions because beekeepers periodically rely on them for expert advice. These associations are often linked to silvicultural companies, which frequently need to implement environmental compensation activities, such as promoting honey production in planted forests and agroforestry systems. These companies are also positively viewed by beekeepers, as they trained and provided the initial equipment to implement beekeeping to most beekeepers. Few beekeepers mentioned NGOs as either potential supporters or opponents, which aligns with the absence of NGOs working with beekeepers in MS.

Results of interviews with other stakeholder groups were equivalent to beekeepers' perceptions. Thus, there is no evidence of conflicting interests or disagreements among the groups involved. Furthermore, the interviews revealed nobody benefits from the damages, as the following evidence confirms.

A consulting government body (CSEAP-MS) congregates all stakeholders responsible for production, sale and consumption of honey, ranging from sanitary requirements, permitting to strategically plan the growth of honey production. The open interview data indicate few stakeholders were aware of the problem or only considered it a localized problem. After presenting our interview results to the CSEAP-MS representatives of (CSEAP-MS), they became strong partners in communicating and supporting solutions (see **Supplementary Material 2** for an official letter of support).

Interviews with representatives of beekeeping associations revealed they are aware of that armadillo's damages. Yet, since the problem occurs throughout part of the state (giant armadillo distribution area, **Figure 1**), it never received consideration. As more interviews were conducted, the associations became strong partners in seeking solutions to reduce giant armadillo's damages (see Desbiez et al., 2020b).

The relationship between beekeepers and silvicultural companies arises from Corporate Social Responsibility practices, which encourages companies to lend their lands so that beekeepers can place their hives. In fact, beekeepers most often place their hives on lands belonging to others. While landowners were never concerned with the issue of giant armadillo predation, eucalyptus companies were. Many areas on their properties have been abandoned by beekeepers due to giant armadillo predation because lethal methods such as trapping or poisoning are forbidden by eucalyptus companies in their own lands. For local NGOs, this issue seems to have been ignored.

DISCUSSION

Understanding the intricacies and drivers of the negative interaction between people and wildlife is necessary, before deciding or suggesting management practices and strategies. By applying the conflict analysis framework proposed by Zimmerman et al. (2020), we gained a good insight about the current coexistence scenario involving beekeepers and giant armadillos. The framework allowed us to describe the current situation at what Zimmerman et al. (2020) called *level one conflict*. This means what limits beekeepers and giant armadillos peaceful coexistence is not yet rooted in less visible, more complex and subjective social disagreements between people and groups (Madden and Mcquinn, 2014). Most of the current challenge is shaped by a material dispute: destruction of beehives.

However, our findings also suggest the conflict could escalate to the second level. A close examination of each of the five key areas of analysis indicates resentment toward the damage causing species is beginning. Beekeepers' responses to the NIMBY question, as well as accumulated frustration about previous inneffective measures to prevent damage, are evidences in this regard.

For the first key area—i.e., beekeepers' perceptions toward giant armadillos—we adopted several indicators. Results of the free listing indicated negative terms were less frequently and less saliently associated with giant armadillos. Instead, neutral terms predominated (usually physical attributes), followed by positive aspects. Furthermore, three indicators (i.e., attitudes, perception of species' benefits and understanding of its needs) pointed out that most beekeepers highly value, understand, and appreciate giant armadillos. Although damage associated with wildlife often decreases a species' appreciation (Dickman, 2010), the cost-benefit balance of living with damage-causing wildlife is accounted as influencing people's views about a species (Treves and Bruskotter, 2014). Thus, when beekeepers were asked about giant armadillos without mentioning their potential damages, positive perceptions were more likely.

However, our last indicator to assess perceptions, the NIMBY sentiment (Dear, 1992; Hamazaki and Tanno, 2002), revealed beekeepers' low preference for spatial proximity with giant armadillos. NIMBY term was coined to characterize residents' motivations to protect their turf from the installation of generally undesirable facilities (e.g., incinerators, jails) (Dear, 1992). The concept assumes people often approve of a certain facility and demand its benefits; even so, they are unwilling to pay the costs of hosting it in their backyards (Hamazaki and Tanno, 2002). While giant armadillos may be positively valued, to beekeepers, hosting them locally is undesirable. The unwanted consequence, i.e., giant armadillos' damage to beehives, can be very destructive, therefore affecting beekeepers' livelihoods (Desbiez et al., 2020b). Strong NIMBY sentiment may reflect some resentment toward the species, and often results in antagonism between those who do not need to host and the "host group" (Dear, 1992); in this case, beekeepers coexisting with giant armadillos. Thus, beekeepers' perceptions that living nearby giant armadillos have negative consequences suggest the situation can potentially escalate to the second level, particularly if group disagreements arise (e.g., between beekeepers and wildlife conservationists).

The second key area of analysis suggests the tangible impacts—i.e., losses inflicted by giant armadillos—are at the core of beekeepers' dissatisfaction, in line with level-1 conflicts. This is evidenced by three of our findings.

1. The damage claimed is very salient (i.e., frequent and very destructive). Most interviewees experienced damage by giant armadillos at least once, occasionally with substantial economic losses. A few were even forced to abandon certain honey production locations, due to the extent of giant armadillos' depredation (Desbiez et al., 2020b). Certain conflicts, when closely analyzed, suggest the negative impacts claimed are not always real, but perceived as such (Dickman and Hazzah, 2016). For instance, Maasai inhabitants in Kenya reported high levels of conflict with lions; despite the actual rates of damage were quite low, with <5% of all depredation events attributed to lions (Hazzah et al., 2009). In these human-wildlife scenarios, there are often other factors behind people's dissatisfaction which explain the conflict. In the beekeepers—giant armadillos' case, the negative impact claimed -the destruction of beehives- is real and not attributable to other species.
2. Damage plays an important role in explaining beekeepers' attitudes toward giant armadillos and to their persecution. Beekeepers affected by higher level of losses were more likely to hold negative attitudes toward giant armadillos and more favorable to their persecution. This contrasts with other studies that claim the extent of the damage experienced does not directly correlate with the retaliation intensity or the attitude direction (e.g., Naughton-Treves et al., 2003; Shelley et al., 2011). When damage does not drive stakeholders' attitudes toward species, mitigating damage would not alleviate human retaliation and negative perceptions of a given species. In this case, instead, damage clearly drives stakeholders' attitudes toward giant armadillos.
3. Beekeepers' tolerance to giant armadillo's damage is high, another aspect suggested by Zimmerman et al. (2020) to classify this conflict at the first level. This observation contrasts with the evidence presented in a meta-analysis conducted by Kansky et al. (2014). When investigating the attitudes and tolerance toward four groups of damage-causing mammals (carnivores, ungulates, elephants, primates), the authors found lower values of tolerance for all stakeholders evaluated.

The third and fourth key areas of analysis—i.e., the history of attempts to solve conflictive issues and readiness to find solutions—showed beekeepers' high willingness to adapt management habits to reduce losses, another indicative of a first-level conflict. In our study, almost all beekeepers were willing or very willing to change their bee management practices to avoid hive losses to giant armadillos. In fact, most of them had already voluntarily implemented at least one non-lethal method to prevent giant armadillos' raids in the last 12 months, reporting variable rates of success. This finding indicates their high readiness to attempt loss reduction without resorting to harmful methods to giant armadillos.

However, less than a third of the interviewees were satisfied with the effectiveness of the attempts. Moreover, 41.4% of interviewed beekeepers had previously adopted one or more measures which failed to prevent damages to their beehives. Frustrating attempts can lead to resentment toward the species, and an escalation to level 2 conflict. Frustration can also lead beekeepers to resort to easier to implement lethal strategies, such as poisoning.

The fifth key area of analysis seeks to understand the quality of relationships between other stakeholders involved in the issue. No resentment was identified to any of the third parties, recognized by beekeepers as potential contributors to solving the giant armadillo predation issue. They would, in fact, welcome help from third parties. This is a key area of evaluations in conflict analysis, since mistrust and misunderstandings with third parties can potentially lead to non-compliance and opposition to conservation initiatives (Young et al., 2016; Baynham-Herd, 2020). Interviews with other groups involved corroborated beekeepers' perceptions. All interviewed representatives demonstrated great concern about the issue, once they learned about it, and welcomed the devise of solutions aimed at promoting peaceful coexistence between beekeepers and giant armadillos. This finding also highlights that the challenges of coexistence between giant armadillo and beekeepers arise solely from the negative interactions of the former with the later instead of interactions with other human stakeholders.

Although the conceptual framework adopted here proved very useful to this purpose, there are two points that deserve consideration.

First, the structure of investigation described by the authors is well laid out to investigate the viewpoint of the directly affected group which interacts with wildlife—in our case, beekeepers. The key areas of investigation are clearly and logically focused on exploring this group perceptions. Although the authors highlight the need to evaluate the beliefs of other human groups involved,

the topic is not further explored, giving a false impression that this aspect is secondary.

Second, it has been previously argued, in the conservation literature, that if a given human-wildlife interaction is only framed by negative impacts between wildlife and people, it does not constitute a *conflict*. Thus, perhaps labeling it as *level one conflict* may be inappropriate.

Conservation Implications

Dealing with challenges arising from human-wildlife coexistence requires adapting approaches according to the strength, characteristics and drivers of a given situation. The framework proposed by Zimmerman et al. (2020), and extended with our methods' detailing, allowed us to evaluate beekeepers-giant armadillos' interaction as predominantly shaped by the economic loss of beehives. The main implication of this is that technical solutions aimed at preventing damage and economic losses and/or by providing financial benefits to people negatively affected by wildlife may successfully promote harmonious coexistence. These findings allowed us to devise three conflict mitigation strategies, together with beekeepers, which are more appropriate to reduce hives' predation and thus human-wildlife negative interactions.

Our first strategy was to compile information learned from this study and from field trials on the efficiency of different measures that may prevent giant armadillo's damages to beehives into a guide which explains how beekeepers can coexist with giant armadillos (see https://www.canastrasecolmeias.org.br/guias-e-manuais and Desbiez et al., 2020b).

Yet interventions which increase knowledge are poor proxies for behavioral changes (Kollmuss and Agyeman, 2002) unless people are motivated to do so (they have an interest in changing) and have the capacity to change (Veríssimo et al., 2019). Our results indicate a high percentage of beekeepers are willing to change, whereas less than a third of them were satisfied with current mitigation practices. By evaluating this aspect in the framework, we became aware of the need to transfer knowledge about more efficient methods. To further enhance our impact, we also have hired a beekeeper to act as an extensionist who supports and guide other beekeepers.

However, as said, interventions such as mitigation strategies are effective only when people are able to change, i.e., to implement them (Veríssimo et al., 2019), but beekeepers' do incur on extra time and financial burden to do so (Desbiez et al., 2020b). Thus, there is a concern that even those more motivated may be unable to implement mitigation strategies. As yet we believe this would not be a problem.

One measure often adopted to mitigate the economic burden of more environmentally-friendly production strategies is certification. Certification of wildlife-friendly products has been frequently implemented to promote human-wildlife coexistence (Bogezi et al., 2019). Based on these previous experiences, we have implemented, as a second strategy, a certification process, labeled *Giant Armadillo Friendly Honey*, which should compensate the extra beekeepers' effort through increased access environmentally-aware niche markets and adding value to their products. There are no financial or administrative costs to enroll in the certification system. By signing a contract, beekeepers must agree to a set of norms, which include best practices and the use of efficient mitigation measures to prevent giant armadillo's predation. The certification system has been approved and is supported by CSEAP-MS, responsible for norm enforcement during inspections. A pilot study was conducted to test the certification implementation and to gather feedback from participant beekeepers. As of May 2021, certification is open to any beekeeper in the ecological range of giant armadillos in this MS state. In the future, the idea is to expand the initiative to the rest of the Cerrado and then, perhaps throughout the species' range.

Our last ongoing measure is to create a smartphone application (app) in which beekeepers can register all events related to giant armadillos, such as beehive's attacks. The app will also serve to keep regular communication with beekeepers about mitigation strategies, the species and market aspects for armadillo's friendly honey. This strategy has two potential positive outcomes. Data registered through this approach can contribute to improve mitigation strategies, besides raising beekeepers' awareness. Moreover, the app will likely increase communication with beekeepers and, in doing so, sustain their regular and long-term involvement in providing quick feedback to implemented interventions.

We expect all these approaches will lead, in the long-term, to retaliation eradication. Coexistence between people and wildlife, even when levels of tolerance are high, can quickly deteriorate (Gureja, 2007). Through the giant armadillo friendly certification process, continued engagement with beekeepers, we expect not only to eradicate both beehives' predation and lethal retaliation, but also to turn beekeepers into allies of giant armadillos' conservation because they are regarded as beneficial rather than a nuisance.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by Brazilian National Commission for Research Ethics (CAAE n° 34264820.1.0000.516). The participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

This study was part of MLC Ph.D. thesis who helped design the study and interviews, analyzed the data, and wrote the previous and current version of the document. CM supervised the data analysis, revised the interview protocol, as well as earlier and

current versions of the article. BO conducted interviews and tabulated data. ALJD coordinates the Giant Armadillo's Project, coordinated and participated in part of the interviews, devised and implemented conservation measures, and revised the current version of the article. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

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Supplementary Material 1 | Analyses details.

Supplementary Material 2 | Official letter of support.

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frontiers in Conservation Science

COMMUNITY CASE STUDY
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Using Participatory System Dynamics Modeling to Address Complex Conservation Problems: Tiger Farming as a Case Study

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Conservation practitioners routinely work within complex social-ecological systems to address threats facing biodiversity and to promote positive human-wildlife interactions. Inadequate understanding of the direct and indirect, short- and long-term consequences of decision making within these dynamic systems can lead to misdiagnosed problems and interventions with perverse outcomes, exacerbating conflict. Participatory system dynamics (SD) modeling is a process that encourages stakeholder engagement, synthesizes research and knowledge, increases trust and consensus and improves transdisciplinary collaboration to solve these complex types of problems. Tiger conservation exemplifies a set of interventions in a complex social-ecological system. Wild tigers remain severely threatened by various factors, including habitat constraints, human-wildlife conflict, and persistent consumer demand for their body parts. Opinions differ on whether commercial captive tiger facilities reduce or increase the threat from poaching for trade, resulting in policy conflict among diverse stakeholder groups. This paper explains how we are working with international conservation partners in a virtual environment to utilize a participatory SD modeling approach with the goal of better understanding and promoting coexistence of humans and wild tigers. We highlight a step-by-step process that others might use to apply participatory SD modeling to address similar conservation challenges, building trust and consensus among diverse partners to reduce conflict and improve the efficacy of conservation interventions.

Keywords: human-wildlife conflict, systems thinking, wildlife trade, wildlife farming, participatory modeling

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INTRODUCTION

To navigate complex social-ecological systems and promote coexistence with wildlife, researchers and practitioners must focus on knowledge generation while increasing access to and use of information that already exists. Inadequate understanding of systems can lead to misdiagnosed problems and unintended outcomes (Larrosa et al., 2016). These misdiagnoses often create or exacerbate human-wildlife conflict (Hübschle, 2017). Conflict mitigation interventions typically focus on tangible disputes (e.g., livestock depredation and retaliatory killings, illegal poaching) without addressing root causes of problems such as inequitable social relationships and processes (Madden and McQuinn, 2014; Baynham-Herd et al., 2018). While a wealth of information about

these social relationships and processes exists, this knowledge often remains on the periphery of decision-making that impacts wildlife management (Bennett et al., 2017). Interventions that neglect to consider social and political context, such as a singular "war on poachers" are therefore unlikely to succeed. In fact, such interventions may inadvertently fuel social-cultural tensions and subsequent conservation-related conflict (Brashares et al., 2014; Challender and MacMillan, 2014). Efforts to reduce human-wildlife conflict, especially for controversial carnivore species like tigers, rarely address these issues (Krafte Holland et al., 2018).

Madden and McQuinn (2014) argue that "conservation efforts would benefit from improved capacity and resources for understanding and transforming the complex drivers of deep-rooted social conflicts impacting wildlife conservation and management actions" (p. 104). Numerous scholars have made a strong case that biodiversity conservation is ultimately a social and political process (Brechin et al., 2002; Lele et al., 2010; Montgomery et al., 2020). Yet, despite a growing body of research focused on the social component of social-ecological systems (Ban et al., 2013), including systems with carnivore species such as tigers (Torri, 2011; Struebig et al., 2018), efforts to apply this research by re-conceptualizing and adapting conventional conservation approaches have been slow (Bennett et al., 2017). As recognition of these challenges grows, the key ingredients for change are already present. Leveraging them might simply require a shift toward systems-level thinking and adaptation.

TOWARD A SYSTEMS APPROACH

Conservation practitioners often acknowledge the complexity of the systems in which they operate, which span a wide array of habitats, stakeholder groups, communities, sectors, and political boundaries. Most adapt and respond to these systems, altering interventions based on data and experience to better achieve their goals. Yet dynamic social-ecological systems make it very difficult to grasp the long-term implications of management actions due to delays between cause and effect (Kim and Senge, 1994). Implementation itself may also change the nature of the problem, influencing the success of the solution (Game et al., 2014; Larrosa et al., 2016). New tools and approaches are needed to advance understanding of systems and build capacity for action (Mahajan et al., 2019).

To advance understanding of systems, researchers have employed modeling approaches such as bayesian belief networks (Bennett et al., 2021), agent-based modeling, social network analysis, and system dynamics (Frerichs et al., 2016). There are also approaches that focus more on planning to help managers improve decision making and outcomes, such as structured decision making (Gregory et al., 2012) and the Conservation Standards (CMP, 2020). For example, use of the Conservation Standards provides a number of benefits, including identifying potential interventions, clarifying theories of change and increasing collaboration; however, it does not explicitly incorporate system behavior such as feedbacks, non-linear behavior or the consequences of time delays. With such a diversity of tools that could be used for understanding complex systems (Voinov et al., 2018), it can be difficult for managers to know where to start.

The field of System Dynamics (SD) began in the early 1960s to better understand complex human and industrial dynamics (Forrester, 1971). Today, SD is used to inform decision making and policy in fields such as business (Ford, 1997; Sterman, 2000), health (Frerichs et al., 2016; Currie et al., 2018), social work (Trani et al., 2016; Appel et al., 2019; Fowler et al., 2019), and agriculture and natural resource management (Ford, 1999; Stave, 2010; Turner et al., 2016; Kopainsky et al., 2017). It has even been applied to species such as sage grouse (Beall and Zeoli, 2008), African penguins (Weller et al., 2014) and grizzly and spectacled bears (Faust et al., 2004).

While not suitable in all cases, SD offers a number of strengths in helping to understand the dynamic behavior of complex systems and test assumptions of different actions and policies with a focus on solving problems (Forrester, 1994; Sterman, 2000). SD traditionally uses two main modeling types: qualitative causal loop diagrams (CLDs) and quantitative simulation models. CLD's identify relationships and feedback mechanisms between elements. Simulation models incorporate the elements of a CLD into a stock-flow structure, where stocks represent what is accumulating in a system (e.g., number of tigers) and flows represent rates of change (e.g., birth or death rate). Structural (e.g. connections between elements) as well as numerical data are incorporated into simulation models to generate endogenous behavior over time under changing conditions and policy interventions. While not meant for predicting or forecasting, simulation models make it easier to explore the potential implications of changing conditions and selected policy interventions on system behavior (Sterman, 2000). SD simulation models run quickly and do not require high computing power; the approach is also particularly useful in environments where quantitative data is scarce and integration of qualitative data (e.g. expert opinion) could be used to address knowledge gaps (Luna-Reyes and Andersen, 2003; Gallagher et al., 2020).

SD is ideally used in participatory planning processes where it can support the negotiation of a shared understanding of a dynamic problem (Vennix, 1999). It facilitates exchange of ideas among participants (Sedlacko et al., 2014) and effectively integrates existing scientific research with local knowledge (Beall and Zeoli, 2008; Stave, 2010; Rouwette et al., 2016). Co-creating SD models necessitates turning implicit into explicit knowledge, so that participants are learning from each other and the model itself (Kopainsky et al., 2017). This also encourages participant ownership of the model and greater support of outputs to address the problem. Model creation can provide a laboratory for a group to examine policies and to visualize potential impacts of actions over time (Forrester, 1994; Sterman, 2000). This is an especially important benefit when working with endangered species or sensitive environments, where physical experiments are not always possible (Sterman et al., 2013; Turner, 2020). In addition to insights from the model, the model building process can increase the social capital of a group (Davies et al., 2015), strengthen relationships and improve communication (Beall and Zeoli, 2008; Stave, 2010). Although

there are several terms used for conducting SD modeling with stakeholders, we use the term participatory SD modeling in this paper.

Despite calls to increase the overall use of models in decision making, resistance may persist for several reasons. Primary concerns include lack of transparency regarding model-building and outputs and weak communication between modelers and practitioners (Addison et al., 2013). Participatory SD modeling offers several advantages since models are designed to be built with stakeholders, using the language of people working on the chosen problem. The visual nature of the modeling software is more accessible to a lay audience, and easy-to-use interfaces help minimize technical barriers between modelers and the modeling groups (Sterman, 1994). Although the value of participatory SD is well-documented (Rouwette et al., 2011; Scott et al., 2016b; Andersen et al., 2017), adoption of this approach to address complex conservation problems has been slow. The time required of participants in the short term (Stave et al., 2019) and the need for a competent modeler and facilitator to coordinate the process (Andersen et al., 1997) are major barriers to adoption. More research and guidance are needed to help conservation practitioners explore the potential value of the participatory SD modeling approach.

This paper explores how a participatory SD modeling process can be used to address a particularly complex problem: conservation of wild tigers.

CONSERVATION CONTEXT: IMPACTS OF TIGER FARMING ON WILD TIGER POPULATIONS

Approximately 3,900 tigers remain in the wild worldwide (World Wildlife Fund, 2021), and they are found in <7% of their original global range (Dinerstein et al., 2007). Wild tiger populations are found in up to 13 countries: Bangladesh, Bhutan, Cambodia, China, India, Indonesia, Lao PDR, Malaysia, Myanmar, Nepal, Russia, and Thailand (Goodrich et al., 2015), although this list includes several countries where wild tiger may be functionally extint (EIA, 2017; Rasphone et al., 2019). The continued survival of tigers depends on a complex set of ecological, economic, and social factors across local and global scales. Because tigers need sufficient lands to roam where they can find adequate prey and live largely undisturbed by people, some experts believe that conservation efforts should focus on law enforcement and protection of habitat and corridors in and around key protected areas (Walston et al., 2010). In areas where tiger conservation succeeds and numbers grow, tigers increasingly come into conflict with growing rural human populations, threatening people and their livestock and potentially increasing revenge killings (Carter et al., 2014; Struebig et al., 2018).

In addition to the interrelated processes of human encroachment, habitat and prey loss, and human-tiger conflict, a persistent consumer market for tiger parts and products economically incentivizes poaching and makes the conservation of wild tigers even more challenging (Wong, 2016). In fact, there is growing consensus that the most urgent threat to wild tigers is poaching (Dinerstein et al., 2007; Chapron et al., 2008). International consumer demand for tiger parts (bones, hides, teeth, etc.) constitutes a major potential threat to wild tigers (Goodrich et al., 2015). Tiger parts are valued across Asia for their perceived health benefits and may confer status and wealth (Goodrich et al., 2015; EIA, 2017). As the species becomes rarer, illegal harvesting and trade in body parts are likely to increase alongside rising market values.

Reduction of the threat of poaching is difficult because tiger poaching crosses multiple countries with different cultures, laws, and policies, and it is influenced by complicated market behaviors (e.g., consumer demand for tiger parts) amidst a growing human population (Sharma et al., 2014). A feedback loop of inter-related increasing scarcity and rising prices can lead to a phenomenon termed the anthropogenic allee effect (Courchamp et al., 2006), which can drive a species to extinction or keep a population low. Under these conditions, drawing attention to the rarity of the species through a demand-reduction program can have the perverse effect of stimulating poaching (Courchamp et al., 2006; Hall et al., 2008). Poaching also appears to have a non-linear relationship with tiger survival, indicating that there are thresholds where even steady rates of poaching could suddenly cause an extinction risk to a tiger population (Kenney et al., 1995). Adding complexity, the tiger trade also potentially threatens other big cats around the world, as body parts from other species such as lions are being traded in the tiger parts market (Williams, 2015; Williams et al., 2017; Villalva and Moracho, 2019; Coals et al., 2020). To mitigate poaching, some have suggested that market demand for tiger products should be supplied from captive sources (Jiang et al., 2007), but this proposal is contested (Gratwicke et al., 2008).

As of 2017, at least 7,000 tigers were estimated to be held in captive facilities (hereafter "tiger farms") across Asia, catering to growing demands for various products ranging from tiger body parts and derivatives to live tigers used for tourist attractions (EIA, 2017). The global captive tiger population is larger, with ∼5,000 captive tigers in the United States alone (World Wildlife Fund, 2020). Many conservation organizations would like to see this practice end, but the potential impacts of closures of farming operations for species are not entirely clear or without risk (Kirkpatrick and Emerton, 2010; 't Sas-Rolfes, 2010). For the purpose of this study, we define a tiger farm as "a facility that keeps or breeds tigers in captivity with an intent (or reasonable probability) of supplying or directly engaging in the commercial trade in tigers or tiger products, be they body parts or derivatives. The application of this definition is not limited by the stated purpose of such facilities."

As Asian economies grow, so might consumer demand for wildlife products such as skins and bones of tigers (Linkie et al., 2018). In one case, researchers found that nearly half (43%) of survey respondents in China (one of the largest consumer markets) had consumed a product that contained tiger parts (Gratwicke et al., 2008). There is uncertainty over the preference consumers may have for wild vs. farmed tiger products (Coals et al., 2020; Hinsley and 't Sas-Rolfes, 2020), with wild tigers possibly being prized more for their power and strength (EIA, 2017). Stronger preferences for wild vs. farmed animal parts have

been reported for other species, such as bears farmed for bear bile (Dutton et al., 2011), but these preferences are dynamic and can shift based on access and availabilty (Davis et al., 2021; Rizzolo, 2021). This uncertainty raises questions about the relationship between tiger farms and demand for tiger parts and products (Song and Yao, 2021).

The challenge of enforcing global wildlife trade under the Convention on International Trade in Endangered Species (CITES) (Challender et al., 2015), combined with limited capacity to combat poachers, has led to some researchers to support limited tiger farming (Abbott and van Kooten, 2011). The argument for legalized tiger farming proposes that increasing the supply of parts will suppress the market price of illegally harvested tiger products (Abbott and van Kooten, 2011). While the demand for tiger parts would persist, diminishing financial incentives for illegally harvesting wild tigers could deter poaching. Based on this hypothesis, some scientists have advocated for humane, renewable harvest and legal trade of other endangered wildlife species facing similar predicaments—such as African rhinos (Biggs et al., 2013).

Conversely, many argue that farming tigers and facilitating the use of their parts for a consumer market fuels market demand and complicates enforcement efforts to reduce wild tiger poaching (Gratwicke et al., 2008; EIA, 2017). According to this argument, the presumed benefits of legal supply might be undermined by imperfections in the tiger parts market, including dominance of a small number of producers controlling prices, the luxury status of tiger parts, and the relatively high expense of farming tigers (Kirkpatrick and Emerton, 2010). Legal markets for farmed tiger products might also lead to greater social acceptability of the product, thereby suppressing a stigma effect considered necessary to prevent unsustainable demand levels (Fischer, 2004; Rizzolo, 2021).

Considering the growth in tiger farms and potential demand for tiger parts globally, many scientists and conservation managers are seeking to better understand the impacts of tiger farming on wild tiger populations. The complex dynamics surrounding tiger farms highlight the need for holistic, systems-based approaches to understand their full impact on wild tiger conservation (Rizzolo, 2021). Understanding complex systems such as those impacting tiger conservation efforts is exceptionally difficult (de Vos et al., 2019), but remains a global priority in conservation science. Participatory SD modeling offers a unique opportunity to understand the problems related to tiger farms and to evaluate the efficacy of proposed interventions.

Below, we describe the development and implementation of a participatory SD modeling process designed to explore the impact of tiger farming on wild tiger populations.

APPLYING THE PARTICIPATORY SYSTEM DYNAMICS MODELING PROCESS

There is a tendency when applying SD modeling to focus on what is perceived to be the final product: the model itself. Although the model can be an important decision-making tool, it is often not the most valuable outcome. The process itself is what creates an opportunity for conflict transformation (Madden and McQuinn, 2014). In the following sections, we discuss the primary steps in a participatory SD process and describe how we are currently applying them to improve understanding of the impact of tiger farming. Because our model building efforts are ongoing, the outcomes are not yet known and some aspects of our participatory approach continue to evolve. However, we have already learned multiple lessons that could help to inform participatory SD modeling in other contexts. Using tiger farming as a case study, the framework outlined in this paper illustrates how similar participatory SD approaches might be designed and implemented to build knowledge, trust and consensus among conservation partners with the goal of improving future conservation interventions.

Like any complex system, this process is not linear. At each step new information is learned and the identified problem may change, along with system components. **Figure 1** depicts our participatory SD process in action (adapted from Beall and Ford, 2010). Key steps in **Figure 1** are described in more detail below. Our process draws from many earlier examples of participatory modeling (Vennix et al., 1990; Sterman, 2000; Beall and Zeoli, 2008; Beall and Ford, 2010; Stave, 2010; Hovmand et al., 2012; Homer, 2019; Wilkerson et al., 2020), synthesizing and adapting these based on participant feedback and study context.

Setup and Design

The first step in a participatory SD project involves ensuring the right people are involved and that the process is tailored to match the scope of the problem. To develop a robust understanding of a complex problem, participants should bring diverse perspectives, knowledge, and expertise. This includes people who may not agree about a problem, its causes, or potential solutions. It is also important consider who is making policy and management decisions and involve these key actors in the process, if possible. This helps generate a model that is comprehensive, valuable to the individuals participating, and supported by leaders. Who participates also depends on the scope of the project (i.e., relevant geographic area, number of organizations or communities involved) and whether the model building will be done in person or virtually. There are benefits to convening in-person, however virtual platforms (e.g., Zoom) can engage more voices across a wider geographic area at a lower cost (Wilkerson et al., 2020). For either setting, group (or sub-group) size should be structured to make sure everyone can participate fully. Other factors to consider when designing the process include the experience level of facilitators and modelers, funding, and time available for both participants and facilitators/modelers.

Our tiger project was initiated by one organization (an international conservation NGO) starting in 2019, but the desire for diverse perspectives led to the creation of a four-person advisory group, each from varying backgrounds, perspectives, and organizations. This advisory group co-created the process with the research team, then selected and invited the rest of the participants. Throughout 2020, we devoted significant time to building understanding of the project within the advisory group and building trust

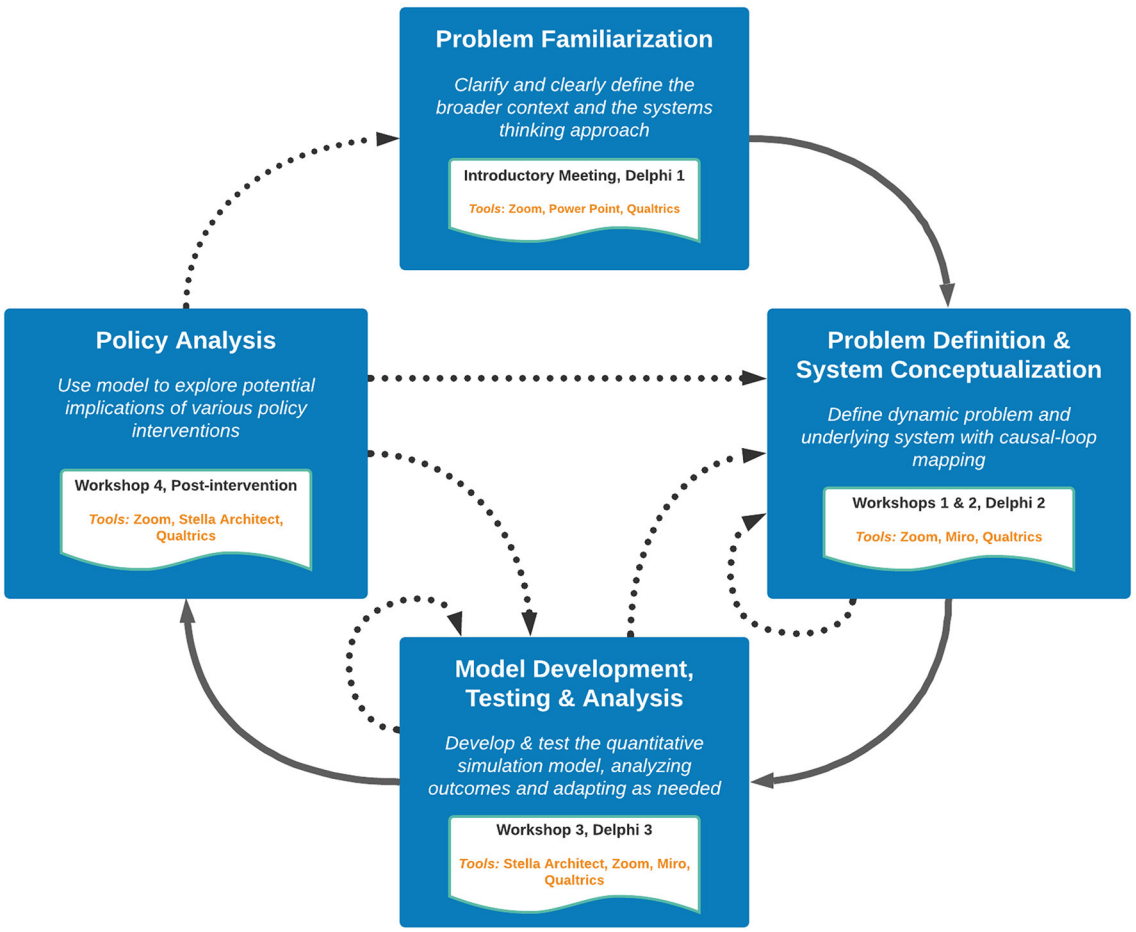


FIGURE 1 | Diagram of the participatory system dynamics modeling process, highlighting key steps in the cyclical process and the activities and virtual tools that might be utilized at each stage. Adapted from Beall and Ford (2010). Solid lines depict direct paths as the stages progresses; dotted lines depict feedback loops and evolving conceptualizations that are a key component of the modeling process. The four workshop structure depicted here can be adapted based on groups' needs, availability, and problem complexity.

among group members. Extensive conversations helped us reach consensus about which participants to invite, ensuring diverse perspectives regarding the costs and benefits of tiger farms while maintaining manageable group size for coordination purposes. To date, our participants include over 50 people spread across conservation NGOs (32 people from 20 different organizations), governmental or intergovernmental institutions (four people), research organizations or institutions (20 people), consultants (five people), and law enforcement (four people) (with participants able to identify multiple sectors). Participants live across Europe, Asia, North America and Africa. Expertise varies among participants, with self-reported knowledge being highest in wildlife trade (76% participants), law-enforcement and anti-poaching (40%), tiger farming in Asia (38%), and farming of non-tiger species (30%). Less than 20% of participants reported high confidence in systems thinking or participatory modeling, demonstrating that this process was relatively novel for most of these conservation practitioners.

Given the global network of experts involved, the costs and logistics made in-person meetings prohibitive (with challenges accentuated by the COVID-19 pandemic); thus, we made an early choice to adopt a completely virtual process. To operate in a virtual environment, the research team needed to learn new tools to be utilized at different stages (**Figure 1**). Recognizing not everyone could (or would) be interested in participating directly in the participatory modeling itself, we created two main groups: a modeling group (including the advisory group) and a consultation group (**Table 1**). To incorporate information from such a large group into the model and to support consensus building among the entire group in a virtual format, we decided to integrate a modified Delphi process into participatory SD modeling. The Delphi technique has been widely used for consensus building about topics ranging from program planning to policy development (Hsu and Sandford, 2007). It utilizes rounds of anonymous questionnaires to explore assumptions, illuminate diverse views, develop a range of possible alternatives, and to educate respondents about complex aspects of a topic.

TABLE 1 | Roles and responsibilities for different stakeholder groups engaged in the participatory system dynamics (SD) modeling process designed to improve wild tiger conservation.

Group	Roles	Important considerations
Research team (2–5 people)	Small group of senior and junior system dynamists who lead model building and facilitation effort; assistance in managing workshops provided by additional researchers.	Modelers and facilitators must act (as much as possible) as honest brokers in facilitating the group process. Modelers are adept at their practice, but they are not subject matter experts (and may be perceived as objective, third-party mediators). Ideally, the modeler and facilitator are separate people.
Advisory Group (3–5 people)	Small, diverse group of experts who co-lead the process with the research team and join the Modeling Group for all modeling workshops.	Members of this group should bring different perspectives to the table. In addition to advising the research team, this group plays the critical role of identifying and inviting appropriate participants for other groups.
Modeling Group (up to 20 people)	Group of experts who, along with the Advisory Group, build the model with the Research team.	Participants should be chosen from different organizations, geographies, and sectors (law enforcement, ecology, wildlife trade, etc.). Ideal candidates have interest, sufficient time, and willingness to collaborate constructively through differences. Group size should be limited to keep workshops manageable, and may be split to facilitate scheduling across multiple international time zones.
Consultation Group (20 or more people)	Additional experts who are invited to contribute to the model through questionnaires and information gathering exercises throughout the process.	Participants include all experts whose input is relevant and important to include. Considerations for selecting individuals for this group might include expertise, organizational affiliation, and time available to devote to the project.

Similar group structure could be used in other participatory SD modeling efforts.

After each round, respondents review summarized responses and highlight areas of disagreement (Hsu and Sandford, 2007), as well as additional questions informed by the modeling (Vennix et al., 1990).

Introduction and Problem Familiarization

Since many participants may not be familiar with systems thinking or the participatory SD process, it is critical to provide a road map to illustrate where the modeling process is going. This overview should include a basic review of systems concepts, such as the definition of a system and the concepts of reinforcing and balancing feedback loops and stocks (what is accumulating or declining) and flows (the rate of change). The introduction should also lay out the modeling process timeline, and show examples of what a model looks like to give participants an idea of where the process will end up. Example models should be relevant to participants, but unrelated to the conservation problem being tackled (Beall and Zeoli, 2008). Finally, it is important to get the group talking about the problem they want to address and to begin working toward defining that problem. The amount of time or focus this takes depends on the particular group, the nature of the problem, and how much clarity and agreement already exists.

The first step in our modeling process was an introductory meeting with all participants following official invitations. The meeting covered the history of the project, introductions, and an overview of the overall process. Basic systems concepts were introduced through real-world hypothetical examples. We used the iceberg model (Senge, 1990) to show that observable events, the tip of the iceberg, are part of larger patterns of change caused by unobservable relationships between elements in a system (also called "system structure"). These are further created and shaped by mental models at the bottom of the iceberg. Changing system structure and mental models produce long term change (Senge, 1990). A demonstration of a simple working dynamic simulation model provided participants with a vision of the end result of their efforts. The meeting ended with a group brainstorm around the broader issue of tiger conservation and the greatest concerns related to tiger farming (**Figure 1**, Problem Familiarization). The most important problems participants identified fell into the following major categories:

- Demand for tiger parts and products (or understanding drivers of demand)
- Lack of understanding the connections between wild tigers and tiger farms
- Market dynamics (price, supply of parts, diversity of products and consumers, etc.)
- Consumer behavior change
- Trade and criminality
- Governance and regulation
- Law enforcement

Setting expectations for participants about the importance of integrating diverse perspectives, including those of potential adversaries, was a key element of the first few meetings. We focused on creating an atmosphere of trying to understand the problem and not to debate specific positions.

Problem Definition and System Conceptualization

Modeling a system without a boundary or a clear focus would produce a model that was unnecessarily, and maybe impossibly, complex and impractical (Sterman, 2000). For this reason, the next step in the modeling process is defining the dynamic problem the group wants to address. A dynamic problem is composed of multiple variables that are changing over time (Homer, 2019), such as a declining wild tiger population, increasing demand for tiger products, and increasing tiger farms. Getting clarity on the problem can be one of the most challenging parts of the process, and may be revisited multiple times as knowledge increases (Mashayekhi and Ghili, 2012). Once the problem has been defined, then the system surrounding that problem can be conceptualized. A qualitative model known as a causal-loop diagram (CLD) is iteratively built based on expert judgment and opinion, followed by reflection about the problem and the system elements. The CLD is later used as the foundation of the simulation model. Approaches to eliciting this initial model vary, but efforts such as Scriptapedia (Hovmand et al., 2012) and the Online System Dynamics Collaborative (https://onlinesd.w.uib.no/) provide tested facilitation scripts to get started.

We are using a combination of questionnaires (that include an adapted Delphi process) and small group workshops to develop consensus around the problem and system components. We began with a pre-modeling survey that contained two open-ended questions focused on understanding the potential impacts of tiger farms on the illegal tiger trade and tiger conservation: (1) what is the most important problem to address within this system? and (2) what factors are contributing to this problem? Answers to these initial questions were summarized and used to inform discussions in the introductory meeting with the modeling group (See introduction and problem familiarization above).

After collecting and synthesizing this information, we hosted a longer workshop to begin the collaborative system conceptualization process. We did this with variable elicitation and behavior over time exercises adapted to the *Miro* online platform (https://miro.com) in Workshop #1. Participant-defined variables, along with relevant behavior over time graphs describing how variables have shifted in the past and predicting how they might shift in the future, were then worked into a CLD. The CLD was modified to include key stocks and flows important in the quantitative model (**Figure 2**) (Homer, 2019). The model was further developed in Workshop #2. Examples of key components of this model include: farmed tiger populations and farm capacity, connections between farmed tiger mortality and sales of products, consumer demand for products and factors that influence demand, and wild tiger population dynamics. Between workshops, the modeling team worked to refine the model and incorporate additional input through one-on-one conversations. When this paper was written, our modeling effort remained in this stage. The following sections outline next steps that could be taken in this, or any other, participatory SD modeling process.

Model Development, Testing and Analysis

Once the initial qualitative CLD is developed around the problem, it is transferred to SD modeling software for creation of a dynamic simulation model. Popular software programs used for this purpose include *Stella* (https://www.iseesystems.com/store/products/) or *Vensim* (https://vensim.com). With this transition, additional questions as well as gaps in logic and knowledge become apparent, and changes to the problem definition or system components are common. Developing, testing and analyzing the quantitative simulation model happens through frequent dialogue between the research team and participants in the modeling group. As the modeler creates the model, they seek input and approval from the modeling group to refine overall system structure and to ensure necessary data are included. Data may include peer reviewed literature, public or private datasets, and local or expert knowledge. Not every relationship and variable within the system of interest will be captured by the simulation model. The focus of SD model building is to build as simple an explanation for the underlying historic behavior as possible. It is impossible to capture all relationships, but this is often unnecessary for understanding the major endogenous influences of problem behavior. As the model is being built, and before it is finalized, it should go through rigorous testing including structure and parameter confirmation, extreme condition testing, and sensitivity analysis (Forrester, 1980; Sterman, 2000).

For this study, we are using *Stella Architect* software for the simulation model and complementing this with *Miro* as a collaborative space for model development. Once the qualitative CLD is sufficiently complete in the previous step, the draft simulation model will be created in *Stella* (see simplified example in **Figure 3**). This model will be built sector by sector, starting with tigers in farms, then demand (and purchasing) of wild and farmed products, and finally linking to poaching of wild tigers. The model draft goes through a model review exercise (https://en.wikibooks.org/wiki/Scriptapedia/Model_Review), after which it is further refined by the modeler. After initial system structure has been determined, the second-round questionnaire using the Delphi process will ask stakeholders to review summarized feedback from round one and the logical integrity of the model, provide input into model parameters, and share additional relevant data sources. Information collected will then be summarized, shared with participants (in Workshop #3 and through another round of Delphi questionnaires in analysis of policy interventions) and used to revise the initial model presented in Workshop #3. In addition to these activities, one-on-one meetings will take place between the modeler and participants to answer questions as they arise.

Analysis of Policy Interventions

One of the primary benefits of an SD model is that it can be used as an experimental platform to explore and evaluate the potential implications of policy interventions (Stave, 2010; Sterman et al., 2013; Turner, 2020). Once the model has been validated and can approximate historical behavior, policy interventions can be added and a more user-friendly interface can be built to

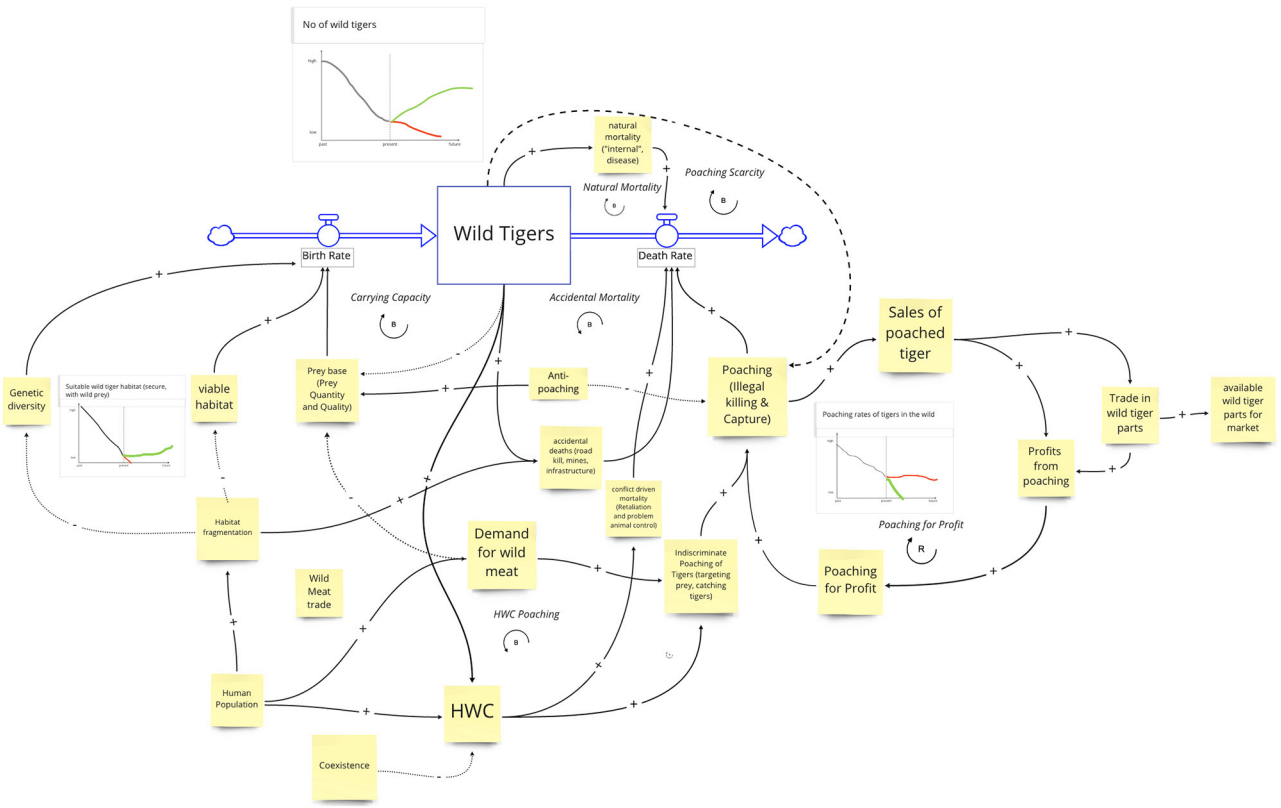


FIGURE 2 | Example of causal-loop diagram (with stock-flow structure added) of wild tiger population developed in *Miro* during Workshops 1 & 2 of the participatory SD modeling process. Arrows depict causal relationships, with solid arrows representing a positive relationship and dotted representing negative relationships.

help stakeholders interact with the model and discern the impacts of various policy options. An example demonstrating the potential of an SD simulation interface is the C-ROADS climate simulation model from Climate Interactive (Sterman et al., 2012; https://www.climateinteractive.org/tools/c-roads/). Stakeholders involved in the participatory SD process choose the policy interventions to be tested, in collaboration with the modeler.

For our case study, the ability to test potential implications of policies related to tiger farms is a main focus. Once the final draft of the model is ready for scrutiny, a final round in the Delphi process will create an opportunity for the full participant group to review summarized results earlier surveys, to provide input into key components and results of the model, and to suggest priority policy interventions. Input from the Delphi questionnaire will be summarized, reported back to the modeling group, and used to add policy interventions into the model. We anticipate this model could provide an opportunity to explore the potential implications of closing or phasing out tiger farms, or tightening restrictions to the trade or sale of tiger parts and products. Once policies and an interactive interface are added, Workshop #4 will give the modeling group an opportunity to test the model and explore the impacts of different scenarios. To mark the end of this stage and the whole process, a full-project presentation will share SD model results with all participants and provide a forum for reflecting on the process and discussing next steps for policy and practice.

Evaluating the Process, Outputs and Outcomes

A key benefit of participatory SD modeling is its potential impacts on shared knowledge building and social outcomes such as trust, communication, and consensus (Rouwette et al., 2002). Evaluation is necessary to verify outcomes and gather feedback to improve further participatory modeling efforts. Participatory approaches to SD have been evaluated in many cases, generally yielding positive outcomes (Rouwette et al., 2002; Rouwette, 2011; Hovmand et al., 2012; Scott et al., 2016b; Stave et al., 2019). Yet, to determine if the process is achieving desired goals, both output and outcome evaluation are an essential part of any modeling effort.

We are integrating evaluation throughout our modeling process. Following recommendations by Scott et al. (2013) we are employing a pre-post survey model. We adapted survey protocols from literature evaluating other collaborative model-building processes (Rouwette et al., 2007; Scott et al., 2016a, 2017), and we are distributing questionnaires to all groups of participants using *Qualtrics XM* software (https://www.qualtrics.

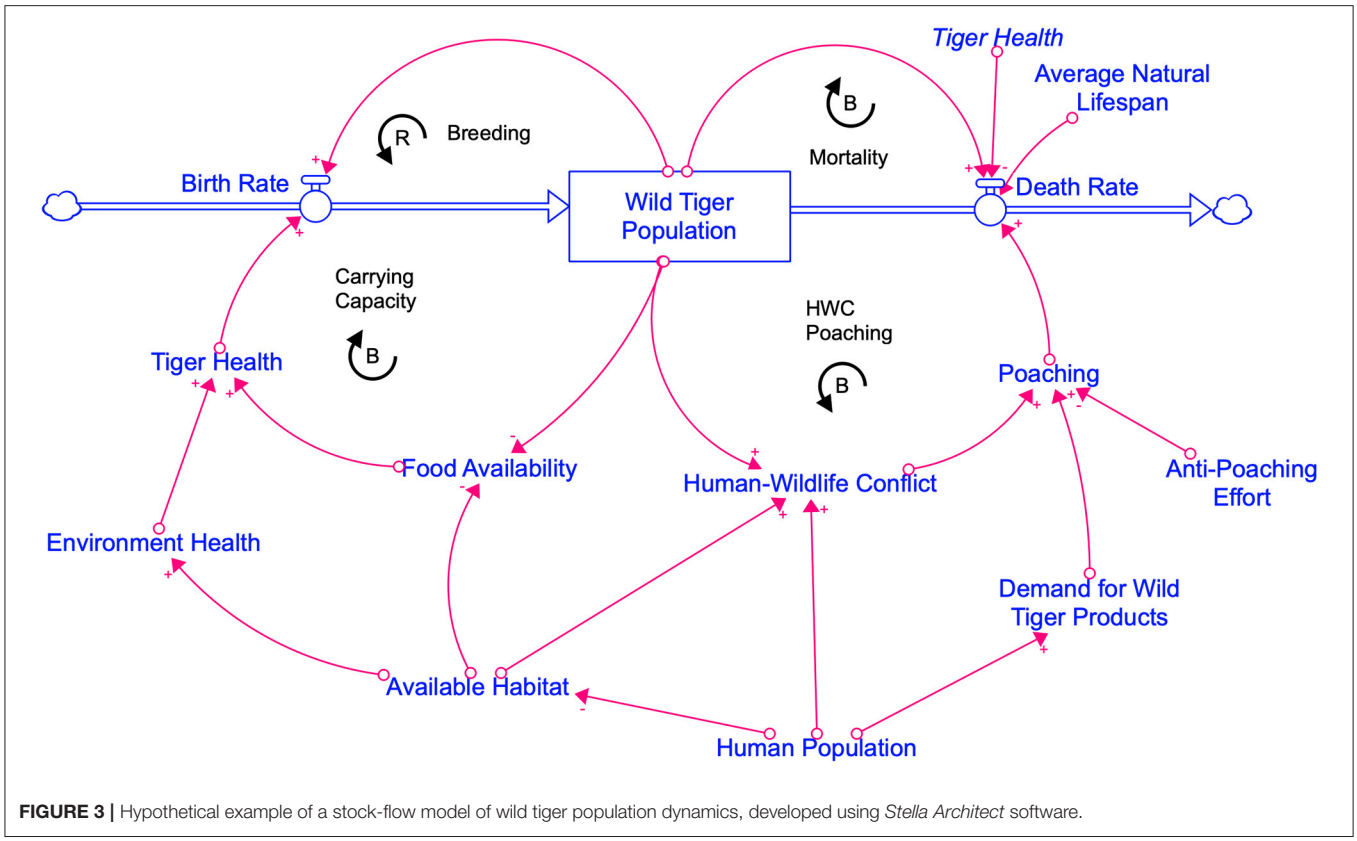


FIGURE 3 | Hypothetical example of a stock-flow model of wild tiger population dynamics, developed using *Stella Architect* software.

com/). The pre-intervention questionnaire, which doubled as the first round of the Delphi process, included questions about participants' areas of expertise, specific perspectives on tiger conservation and tiger farms, and previous experience with systems thinking (see **Supplemental survey instrument**). We also included questions designed to measure key process outcomes such as knowledge and understanding (Rouwette et al., 2016; Scott et al., 2016a), trust (Stern and Coleman, 2014; Basco-Carrera et al., 2017), and consensus and commitment among conservation practitioners (Rouwette, 2011; Scott et al., 2016b; Basco-Carrera et al., 2017). We aim to integrate a post-intervention questionnaire that allows the research team to measure changes in responses from the beginning to the end of the modeling process. This final questionnaire will include additional questions to gather feedback about understanding of the dynamics in the tiger conservation system (i.e., connections between wild tigers, demand for tiger parts and products, and tiger farming), the utility of the final model, and perspectives on how the process itself influenced perceived outcomes such as knowledge, trust, communication and consensus and commitment (i.e., the same outcomes addressed on the pre-intervention questionnaire). Multiple rounds of Delphi questionnaires integrated throughout the process will help us track the evolution of participants' thinking regarding the problem(s) and the complex system surrounding tiger conservation.

IMPLICATIONS FOR CONSERVATION PRACTICE

Conservation practitioners work in complex social-ecological systems to address threats facing biodiversity, reduce conflict, and promote positive human-wildlife interactions. Inadequate understanding of the direct and indirect, as well as short- and long-term, consequences of decision making within these dynamic systems can lead to misdiagnosed problems and interventions with perverse outcomes, exacerbating conflict (Larrosa et al., 2016; Hübschle, 2017). Participatory SD provides an opportunity to minimize these risks through building a more complete shared understanding of a problem and potential implications of interventions. This is achieved while increasing trust and reducing conflict among stakeholders working to tackle these wicked problems. Once created, a simulation model can also be used as an experimental platform that is almost impossible to replicate *in situ* with threatened ecosystems and endangered species.

The process we have outlined in this paper shows how conservation researchers and practitioners can design and implement participatory SD modeling to address a complex problem such as wild tiger conservation. Throughout our ongoing modeling process, we have confronted conflicting perspectives and worked toward shared understandings of the tiger farming problem and its consequences. Through iterative

meetings and conversations combining science with expert knowledge, we are building trust and fostering productive collaboration. As our simulation model progresses, it should yield insights regarding policy interventions that enhance the value of the process for participants. By strategically dissecting the social and political relationships that fuel many conservation conflicts (Madden and McQuinn, 2014), participatory SD processes like ours may be a key step on the path to sustainable coexistence between humans and tigers.

The participatory SD modeling process does not occur without challenges (Addison et al., 2013; Stave et al., 2019). It requires a large time commitment for both the research team and the modeling group. Our study, for example, will have taken multiple years from the initial advisory group meetings to development of the final policy model. The process at minimum requires a competent modeler and, ideally, a facilitator who has experience with systems modeling. Since participatory SD modeling is not common in the conservation world, there is a learning curve for participants to help them understand where the process is going and how to realize its value. Additionally, with a polarized topic such as tiger conservation, it is challenging to find diverse stakeholders willing to participate. Finally, while SD models provide insights that can help to guide management and policy, key decision makers must be willing and able to utilize these tools to initiate action. Strategies for addressing these potential barriers undoubtedly vary by context, but investments of time and resources into systems-based approaches could ultimately lead to long-term changes the way conservation efforts are conceptualized and carried out.

Despite challenges, enthusiasm with our effort to model the impacts of tiger farming on wild tiger conservation remains high. Some participants may not be able to engage in the whole process, but excitement has grown as conversations delve deeper into complex issues and the practical implications and potential policy impacts of the effort become more apparent. Our approach is showing how systems thinking and systems-based approaches can help to address the complex social, economic, political, and ecological problems that threaten the survival of wild tigers. Application of systems thinking could improve coexistence with other species where contentious policy choices are being critically evaluated, such as elephants (Mahajan et al., 2019), rhinos ('t Sas-Rolfes, 2016), and wild horses (BLM, 2020); it could also facilitate understanding of conservation issues that span multiple species and contexts, such as the substitutability of tiger and lion products across the farmed/wild nexus (Coals et al., 2020; Rizzolo, 2021). Additionally, SD approaches create unique opportunities to explore the effects of different property rights or management regimes on wildlife (Wilson et al., 2016). Regardless of geography, focal species, or management context, participatory SD modeling could represent a valuable tool in a conservation practitioner's toolbox to address conflict and improve coexistence with wildlife around the world.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by Institutional Review Board of North Carolina State University (IRB 21209). The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

ER and LL are the principal investigators of the study and oversaw design of the conceptualization of the study, design and implementation of the process (including workshop facilitation), and the writing of this manuscript. BK helped design the participatory SD process, assisted with workshop facilitation, and assisted with manuscript writing. M'tS-R participated in the participatory SD process and assisted with manuscript writing. All authors contributed substantially to the article and approved the submitted version.

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We would like to acknowledge the participants from NGOs and other organizations around the world who continue to contribute to our participatory modeling process as we work collectively to advance tiger conservation.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.696615/full#supplementary-material

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Working Together for Grizzly Bears: A Collaborative Approach to Estimate Population Abundance in Northwest Alberta, Canada

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Grizzly bears are a threatened species in Alberta, Canada, and their conservation and management is guided by a provincial recovery plan. While empirical abundance and densities estimates have been completed for much of the province, empirical data are lacking for the northwest region of Alberta, a 2.8 million hectare area called Bear Management Area 1 (BMA 1). In part, this is due to limited staff capacity and funding to cover a vast geographic area, and a boreal landscape that is difficult to navigate. Using a collaborative approach, a multi-stakeholder working group called the Northwest Grizzly Bear Team (NGBT) was established to represent land use and grizzly bear interests across BMA 1. Collectively, we identified our project objectives using a Theory of Change approach, to articulate our interests and needs, and develop common ground to ultimately leverage human, social, financial and policy resources to implement the project. This included establishing 254 non-invasive genetic hair corral sampling sites across BMA 1, and using spatially explicit capture-recapture models to estimate grizzly bear density. Our results are two-fold: first we describe the process of developing and then operating within a collaborative, multi-stakeholder governance arrangement, and demonstrate how our approach was key to both improving relationships across stakeholders but also delivering on our grizzly bear project objectives; and, secondly we present the first-ever grizzly bear population estimate for BMA 1, including identifying 16 individual bears and estimating density at 0.70 grizzly bears/1,000 km^2-the lowest recorded density of an established grizzly bear population in Alberta. Our results are not only necessary for taking action on one of Alberta's iconic species at risk, but also demonstrate the value and power of collaboration to achieve a conservation goal.

Keywords: conservation, governance, collaboration, management, grizzly bear (*Ursus arctos*), population, theory of change, SECR

INTRODUCTION

Grizzly bears (*Ursus arctos*) are an icon of the North American wilderness and a symbol of both conservation and conflict (Proctor et al., 2018; Hughes and Nielsen, 2019; Hughes et al., 2020b). However, as with other large carnivores, human-caused mortality, including illegal killing and vehicle collisions, as well as implications from habitat alteration are the primary threats to grizzly bears (Alberta Sustainable Resource Development., 2008; Nielsen et al., 2009). While scientific research and applied management is ongoing across much of North America, data deficiencies persist in grizzly bears' more northerly range, including those in the boreal northwest region of Alberta, Canada.

Grizzly bears in Alberta were listed as threatened in 2010 due to their small population size, slow reproductive rate, limited immigration from other populations, and increased habitat alteration (Alberta Sustainable Resource Development., 2008). A recovery plan identified seven demographically-separate bear management areas (BMA) with recovery objectives including the necessity for population estimates to be conducted for each BMA, and addressing human-caused bear mortality through access and attractant management along with educational outreach (Alberta Sustainable Resource Development., 2008). From 2004 through 2010, a series of DNA-based population inventories were completed across Alberta, including testing non-invasive genetic techniques and modeling estimates by Foothills Research Institute in the boreal northwest (Alberta Sustainable Resource Development., 2008; Festa-Bianchet, 2010). Based on these data, alongside habitat modeling and expert opinion, Alberta's estimated grizzly bear population was 691, plus additional bears in portions of Banff and Jasper National Parks (Festa-Bianchet, 2010).

Estimating grizzly bear density in the northwest population unit called Bear Management Area 1 (BMA 1) has remained particularly challenging, given the large and relatively remote geographic area and wetland conditions of this landscape. This is contrasted with other BMAs, which are largely comprised of the Rocky Mountain and Foothills natural regions and increased human density, resulting in more road or trail access into grizzly bear habitat (Alberta Environment Parks, 2020). While problematic for human-caused mortality and habitat fragmentation, this increased linear footprint in other BMAs generally reduces the costs associated with efficiently inventorying bear populations.

Another challenge in BMA 1 are the differing perspectives and experiences people have across the region concerning grizzly bears, which can hinder effective conservation and management efforts. Grizzly bears are a charismatic species valued for their aesthetics as well as ecosystem function, but also a species that poses serious human safety risk and economic costs to peoples' livelihoods (Morehouse and Boyce, 2017; Proctor et al., 2018; Hughes and Nielsen, 2019; Hughes et al., 2020a; Morehouse et al., 2020). People across BMA 1 hold values and cultural identity linked to the concept of "frontiersmen," with their ancestors being hardy pioneers of this harsh boreal landscape (Hughes and Nielsen, 2019). Human-bear relationships are viewed and experienced from the perspective of subsistence lifestyles, generating income, and ensuring human safety, and today this still resonates with many people who call the northwest home (Hughes and Nielsen, 2019). That said, the cultural identity, values and practices in the northwest has conflicted with the provincial government's grizzly bear recovery policy, including how provincial direction governed industrial-scale petroleum and forestry production (Hughes and Nielsen, 2019; Alberta Environment Parks, 2020). Tension between local government and stakeholders across BMA 1 has persisted since the late 1990's (Fullerton, *pers. comms*). This may be related to a lack of trust in grizzly bear science and scientists, or inaccessibility of scientific information and lack of layperson understanding, as well as local perspectives that problem bears were simply "dumped" (i.e., re- or translocated) into BMA 1 thus contributing to human-bear conflict (Hughes and Nielsen, 2019). During this time public reporting of human-grizzly bear interactions was limited, and a "shoot, shovel, and shut up" sentiment was commonly expressed to occur across rural communities in the northwest (Hughes et al., 2020a).

To address challenges associated with a population inventory of BMA 1 grizzly bears and improve local relationships to enable progress on grizzly bear management, we implemented a collaborative approach that engaged representative stakeholders from across this multi-use working landscape (Wondolleck and Yaffee, 2000; Wilson et al., 2017; Hughes et al., 2020a). Collaborative approaches have been used around the world, including to address bear hunting in Romania (Hartel et al., 2019), conflicts with gray wolves in Montana (Wilson et al., 2017), lion conservation in Zimbabwe (Sibanda et al., 2020), and human-bear coexistence in southwestern Alberta (Morehouse et al., 2020). Collaborative approaches are considered effective at bringing different people across multiple disciplines, perspectives, and experiences together to identify and achieve defined outcomes (Yang, 2017; Hartel et al., 2019; Hughes et al., 2020a). Additionally, collaborative approaches enable participants to decentralize decision-making and share power, foster fairness, and improve credibility and trust in project or policy processes (Singleton, 1998; Wondolleck and Yaffee, 2000; Carlsson and Berkes, 2005; Mattson et al., 2006; Clement et al., 2020). Moreover, collaboration can help groups access and leverage shared resources and funding opportunities, as well as foster co-learning amongst participants.

Our paper weaves together our collaborative approach with a multi-stakeholder team to help identify and address data gaps for BMA 1 grizzly bears, the results from a qualitative evaluation of these collaborative efforts, and the results of the grizzly bear population inventory. We also provide considerations for implementing collaborative approaches elsewhere, to help address potentially contentious contexts for the conservation and management of wildlife species.

STUDY AREA

BMA 1 is within a multi-use landscape in the boreal forest of northwest Alberta, adjacent to historical grizzly bear habitat



FIGURE 1 | Provincial map of Alberta with bear management areas (BMA), recovery and support recovery zones (Alberta Environment Parks, 2020).

in British Columbia, and covers ∼41,000 km^2 (Poole et al., 2001; Alberta Environment Parks, 2020). It is comprised of boreal and mixed-wood natural regions with extensive wetland complexes, with only 2% of the land base protected under provincial park designation limiting motorized (vehicle) access (Alberta Environment Parks, 2020; **Figure 1**). Human use across the area includes a history of extensive petroleum developments (i.e., well-sites and pipelines), forestry harvest, electrical transmission, agricultural areas for livestock and crop production, recreational use including hunting, off-highway vehicle enthusiasts, river travel and camping, and small residential communities and farmsteads.

Grizzly bear habitat in BMA 1 is classified either Recovery or Support Zones (Alberta Environment Parks, 2020). The Recovery Zone, an area covering 23,458 km^2 delineated through habitat modeling and expert opinion on bear occurrences, is the focus of our study area (Nielsen et al., 2009). The Recovery Zone identifies where the Alberta government reasonably expects to manage the presence of grizzly bears and reduce human-caused mortality, which has been associated with open road density (Alberta Environment Parks, 2020). The Support Zone (∼18,000 km^2) is intended to allow for grizzly bears to disperse, with management focusing on securing food attractants and teaching bear safety, largely with agricultural landowners and recreationalists. The Alberta BearSmart program (www.alberta.ca/alberta-bearsmart-program-overview.aspx), in existence since 2008, is the primary outreach strategy used to educate the public and address human-bear conflicts.

MATERIALS AND METHODS

Collaborative Planning: the Northwest Grizzly Bear Team

In 2011 government staff, led by the regional manager, coordinated a meeting between local petroleum industry and forestry representatives to identify research and management needs for grizzly bears in BMA 1. This first meeting was an integral step in fostering a collaborative working group in the area. The main topic discussed was how to best address the persistent data gaps on the local grizzly bear population, which is required to meet recovery objectives, as well as discussions on how these data would assist in land use planning, including guiding forest management. However, efforts at the time were hampered by a lack of funding and staffing, coupled with public skepticism, limited local understanding of current scientific information, and lack of trust in government agencies and scientific methods. As a result, a pilot project was initiated between 2012 and 2014 to test the efficacy of bear hair collection procedures and open communication between government and stakeholders. This work helped to share information, seek participation from academic and other scientists, identify funding opportunities, and cooperate with local landowners and industry personnel, specifically to identify bear use areas and rub objects (i.e., trees and power poles that bears rub on as a form of communication) in BMA 1 (Morehouse et al., 2021). These efforts were an important step forward in rebuilding trust and generating enthusiasm across different groups of people in BMA 1. In turn, this became the impetus for formalizing a collaborative multi-stakeholder working group in 2015, called the Northwest Grizzly Bear Team (NGBT; **Table 1**; Wondolleck and Yaffee, 2000; Ansell and Gash, 2008).

Invitations for those interested in voluntarily participating in the newly formed NGBT reflected the suite of different land users and stakeholders across BMA 1. Invited representation included local petroleum industry and forestry representatives, those from the electrical utility sector, government staff, academic and research scientists, agricultural landowners, municipal government representatives, and Indigenous community representatives. Given the types of land use overlapping the grizzly bear recovery zone in BMA 1, participation in the NGBT largely reflected industrial scale natural resource production, with agricultural landowners, municipal government and Indigenous communities declining future participation for various reasons (i.e., perceived relevancy, time commitment,

TABLE 1 | Northwest Grizzly Bear Team composition.

Stakeholder sector	Number of representatives
Forestry	6
Electric company	1
Petroleum	2
Non-profit organizations	3
Government	9
Public at large	1
Total	22

Co-chairs were represented by one Alberta Environment and Parks staff, given the mandate for grizzly bear recovery, and one forest company staff by nomination from other stakeholders.

subject matter expertise) and reasons unknown (i.e., no response to participate despite repeated requests).

In order to ensure common understanding around scope of work, particularly related to legislative and recovery policy requirements, the NGBT developed a terms of reference (TOR). This included expectations set out for member conduct, interpersonal conflict management, and consensus-based decision-making, along with identifying a shared vision, objectives and strategies to achieve objectives, and limitations. To develop the TOR, our discussions followed principles adapted from interactive group decision-making processes, to identify consensus and develop a prioritized list of actions that would resonate with the NGBT (Emerson and Nabatchi, 2015; Huge and Mukherjee, 2017; Mukherjee et al., 2017). Generally this included: (1) defining the management or research questions about BMA 1 grizzly bears; (2) brainstorming on how to address these questions; (3) clarifying and consolidating ideas; and, (4) agreeing upon the top priorities (Huge and Mukherjee, 2017). Based on these discussions, a theory of change (TOC) model was developed to assist in project planning, grant writing, and project evaluation (**Figure 2**; Margoluis et al., 2013; Morehouse et al., 2020).

Theory of Change models have increasingly been used in different conservation contexts to help plan, implement and evaluate projects, given their utility to conceptually illustrate different connections between activities and outcomes (Biggs et al., 2016; Balfour et al., 2019; Sibanda et al., 2020; van Eden et al., 2021). Particularly critical to successfully acquiring substantial funding for this project was collaboratively co-authoring a compelling grant proposal for benefactors, which articulated how investing in our applied project would directly address current political challenges and be valuable for government, industry, the public and grizzly bears.

Lastly, we conducted a summative evaluation of our collaborative governance arrangement to determine the efficacy of this approach and provide recommendations for future efforts (Emerson and Nabatchi, 2015; Robinson et al., 2020). We used a qualitative semi-structured questionnaire asking the NGBT to reflect on and explain their motivations for joining the NGBT, benefits, challenges, outcomes, and future recommendations (**Supplementary Material A**). The questionnaire was developed by the government co-chair with input and review by three members from the NGBT, and shared with the membership for completion. Given the small group size ($n = 22$), we sought to ensure respondent confidentiality and anonymity by using a numerical code for each respondent and clarified how data would be stored and used (Kaiser, 2009; Creswell and Poth, 2017). Using the questions as guiding codes, we identified common themes across the dataset (Guest et al., 2014; Creswell and Poth, 2017). Results were shared back with the NGBT for verification and validation (Creswell and Poth, 2017).

DNA Field Methods

We designed a non-invasive DNA-based (i.e., grizzly bear hair) population inventory to estimate grizzly bear density and abundance using spatially explicit capture-recapture (SECR; **Supplementary Material B**; Boulanger et al., 2004, 2018; Efford and Fewster, 2013; Rovang et al., 2015; Morehouse and Boyce, 2016). We used simulation modeling with the *secrdesign* package (version 2.4.0., Efford, 2016) in R (R version 3.2.5) to design the study area configuration and guide sampling efforts. We used our simulation modeling results to inform our hair trap density and spacing. Results from our simulations indicated that the optimal size of the required sampling grid was dependent on the size of the estimated bear population. To ensure the highest probability of program success, which included in-depth discussions with NGBT members on required staffing and financial resources, we ultimately chose to use a sampling grid of 10 km^2.

In addition to sampling stations across BMA 1, we collaborated with the government of British Columbia's (B.C.) Ministry of Forests, Lands, Natural Resource Operations and Rural Development (FLNRORD) to include 32 sampling stations in the adjacent Taiga Grizzly Bear population unit in British Columbia. We anticipated this would provide important additional sampling effort which would help reduce model uncertainty. Any data collected by B.C. government staff was sent to our team for analysis and reporting.

We selected sampling stations based on habitat features, accessibility, and soliciting expert knowledge from the NGBT members. In total, we installed 222 barbed wire hair snare corrals and 32 rub object stations to non-invasively collect grizzly bear hair samples. Each wire corral was set at ∼4 m in width by stretching four-pronged barbed wire taut around the outside of trees or stakes, at a height of ∼60 cm (Kendall and McKelvey, 2008; Kendall et al., 2009). The same barbed wire was wrapped at ∼6 ft in height around the different rub objects in the sampling grid (Morehouse and Boyce, 2016). Liquid lure made of rancid cow blood and fluids from rotten fish was poured in the center of each wire corral, or splashed on rub objects, and loosely covered with woody debris to protect it from rainfall and drying. All corral and rub object site information was shared with the NGBT so that members could communicate with their field staff to better ensure safety during operations. We also asked petroleum and forest industry staff, as well as agricultural landowners, to report any grizzly bear sightings online or directly to government project staff if and when they occurred, including location information.

To improve model performance, provide variability in spacing between accessible wire corral stations, and increase the total

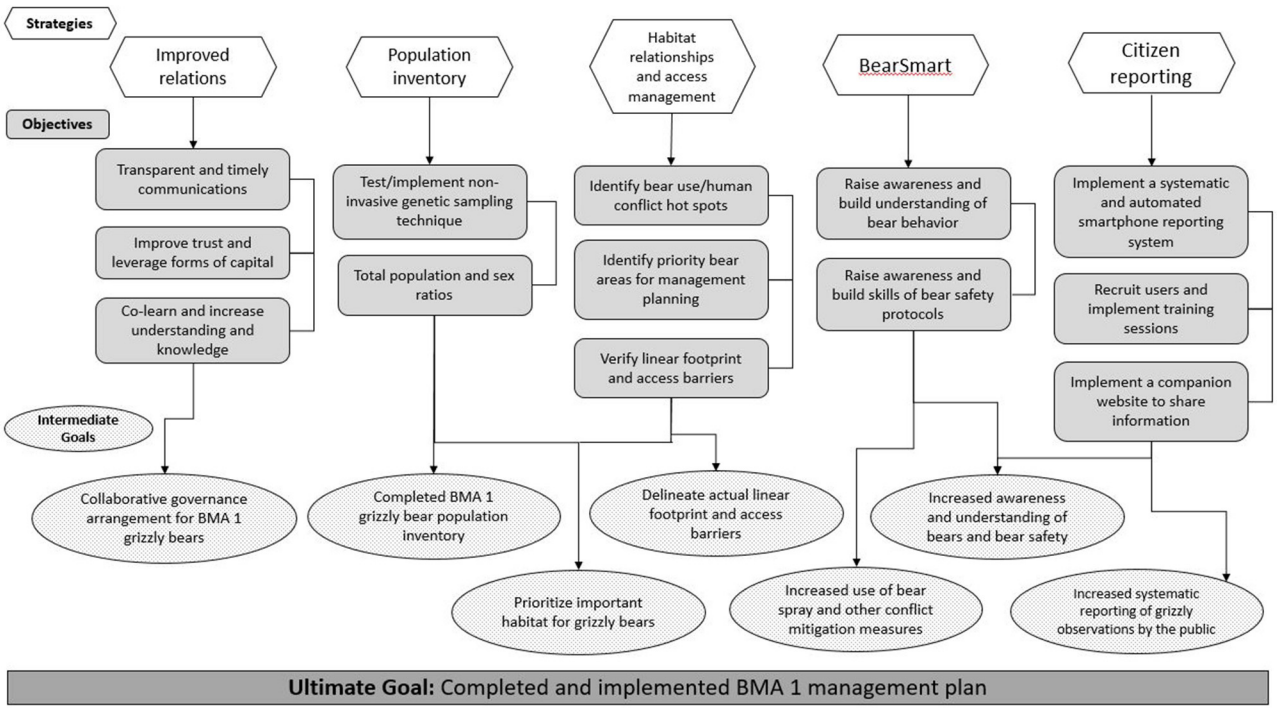


FIGURE 2 | Northwest Grizzly Bear Team's Theory of Change conceptual model.

number of sampling stations without significantly decreasing field crew efficiency, we opportunistically added additional wire corral stations to the grid along existing travel routes (i.e., roads). These stations were predominately focused in the ground-accessible portions of the grid, which may have introduced a slight sampling bias in areas of higher road density.

Each station was visited every 2 weeks (14-day intervals) between May 15 and July 19, 2017, to ensure the integrity and genetic viability of samples present/collected (Stetz et al., 2015; Lamb et al., 2016). All hair on a single barb was considered an independent sample upon each visit, and once collected was stored in a numbered envelope with corresponding site data. The barbed wire corral was then burned with a torch to minimize the possibility of any remaining genetic material contaminating future sampling, and re-lured (Kendall et al., 2009). All hair samples were stored in a dry environment away from sunlight until DNA analysis.

Analysis Methods

We sent all hair samples to Wildlife Genetics International (WGI) in Nelson, B.C. to identify the species (i.e., grizzly bear vs. black bear), sex, and individual identity through an analysis of nuclear DNA which they extracted from the hair follicle (Paetkau, 2003, 2004). Selection and analysis of hair samples for grizzly bears was done using a randomized sub-selection strategy, which included a tiered approach based on hair quality and quantity of guard and underfur hairs (**Supplementary Material C**). In the first tier of sub-sampling, high quality samples with more than one guard hair and 20 or more underfur hairs were selected. If there were not enough high-quality samples to meet these sub-selection rules, marginal samples with one guard hair and 5–19 underfur hairs were selected. Any samples with less hair than the aforementioned thresholds were treated as "inadequate" and were not selected for analysis. One exception was in 36 collection events where all available samples were classified as "inadequate." As a result, the best available sample from each of these collection events was included in analysis at the discretion of WGI. Hair samples were genotyped to eight markers (seven microsatellites plus an additional marker for sex differentiation) using QIAGEN DNeasy Blood and Tissue kits (2021), which enabled reliable differentiation between individuals (Paetkau, 2003, 2004).

We estimated grizzly bear density and abundance using the *secr* package (version 3.1.5, Efford, 2018) in the program R. We used a half-normal detection function and defined the area of integration as a 30 km buffer around the outermost hair snare stations. We did not impose a habitat mask because there were few known non-habitat areas within the study area. Due to limited detection events, density (D) was modeled as a uniform parameter across the study area. Sex-specific models were not possible for similar data limitations. We allowed detection probability (g0) and the spatial scale parameter (σ) to vary as a function of sampling site type (i.e., lured corral or rub object). We compared model performance using Akaike's Information Criterion corrected for small sample sizes (AIC_c) (Burnham and Anderson, 2004). Using our most parsimonious model, we derived grizzly bear abundance for the BMA 1 Recovery Zone.

The NGBT was kept apprised of decisions made and steps taken throughout all stages of analysis, and their input was

solicited and questions answered where and as needed. This step helped alleviate concerns or confusion around analytic methods and prepared members for the forthcoming discussion on project results.

RESULTS

Evaluation of the Northwest Grizzly Bear Team

Fourteen of twenty two questionnaires were completed, including six from forestry representatives, six from government, one from the energy sector, and one from the public member at large. Thirteen of fourteen members indicated their primary motivation to join and participate in the NBGT was a requirement of their employment/position, demonstrating their organization's interest and commitment to grizzly bear recovery, with one respondent indicating it was their personal interest. Over half ($n = 8$) indicated they spent between 5 and 10 h monthly on this project, which was reported to be a reasonable investment of their time, whereas the co-chairs and project science lead indicated spending more than 15 h weekly. This included organizing, coordinating, facilitating and participating in over 15 different meetings, substantial financial administration, numerous different forms of reporting, and field work. Furthermore, the NGBT commissioned the production of an online video called "*Working Together for Grizzly Bears*"[1], through additional fundraising, to broadly and publicly communicate the important collaborative work of this team.

All respondents indicated they had no hesitations in joining this team and reported their curiosity about how this collaborative arrangement would function. Upon reflection, all felt that their participation was meaningful, with comments stating that the teams' professionalism, respectful conduct, cooperative spirit, and knowledge and information exchange were beneficial. When asked if they were satisfied with the outcomes, all but one respondent said yes, with this individual indicating their disappointment in the lack of implementing a BMA 1 plan post-project.

More specifically, the positive outcomes respondents noted included the benefits of collaboration and interactions, trust-building and information exchange (71.4%); the grizzly bear population estimate and verification of BMA 1 linear footprint through industry participation (42.8%); acknowledgment of the importance of the agricultural interface area and related relationship building (14.3%); leveraging funding to achieve project results (14.3%); and, the near perfect safety record (i.e., no major human safety incidences despite risks of remote work in bear country; 7.14%). Further, NGBT members indicated that despite the challenges of collaboration, collectively working together helped them build an understanding of the scientific methods used in grizzly bear population monitoring and the importance and use of scientific data in decision-making (92.8%).

[1]Uploaded by Let's Go Outdoors, 9 April 2019; https://www.youtube.com/watch?v=Gn-bQXUcN6candt=304s.

Reported negative outcomes and concerns included criticism of the lack of senior government officials' commitment to the project, including financially (21.4%); limited participation from the local energy sector, particularly given their influence on the landscape (e.g., linear footprint, bear mortality) as well as lack of engagement by Indigenous or municipal government representatives (21.4%); limited formal recognition of the importance of the NGBT as a model for collaborative governance coupled with concerns regarding support for future BMA1 planning and implementation and ensuring ongoing and active participation in the NGBT (21.4%). One respondent indicated negative outcomes was personally wanting more time to be able to participate in the NGBT.

Respondents also suggested considerations for future efforts, including increasing the NGBT meeting frequency (from bi-monthly scheduling), broader communication to the public about our work, pursuing official government policy direction and support to complete and implement a BMA 1 plan, and increasing engagement with Indigenous communities, agricultural landowners, and municipal government.

Grizzly Bear Density Estimation

We collected a total of 4,208 hair samples during the four sampling periods of our field season. We had 23 detections of 14 unique grizzly bears (12 males, 2 females). This included nine re-detections of grizzly bears from the pilot study, and six movements of individual bears between hair corral sites (**Figure 3**). Of the 23 total detections, 14 occurred at hair corral sites, nine occurred at rub objects (i.e., trees), and all occurred within Alberta. Also of note is that we had 852 detections of 585 individual black bears (333 males, 259 females), with 50 occurring at rub objects and 802 occurring at corral sites.

The top performing model estimated a grizzly bear density in the Recovery Zone of 0.70 grizzly bears/1,000 km^2 [Realized Standard Error (RSE) = 0.349, 95% Confidence interval (CI): 0.36–1.35]. The expected grizzly bear abundance within the BMA 1 Recovery Zone was 16.3 grizzly bears (RSE 0.349, 95% CI: 8.4–31.8).

An additional outcome, and benefit to the project, was the ability to efficiently leverage human and financial capital, including establishing field teams to collect bear hair from the sampling stations, through the NGBT collaboration. Moreover, these field teams also helped foster positive relationships with petroleum industry staff, forestry personnel, and agricultural and other landowners across the study area through informal conversations detailing our project scope and activities. These unanticipated educational opportunities represented an important time for clarifying the logistics and protocols of non-invasive genetic field methods and data uses in applied management, which in turn increased our study's transparency, sparked curiosity in grizzly bear science, and ultimately helped improve government-stakeholder-public relations.

DISCUSSION

We used a collaborative governance approach, including jointly developing Theory of Change (TOC) to guide and evaluate our

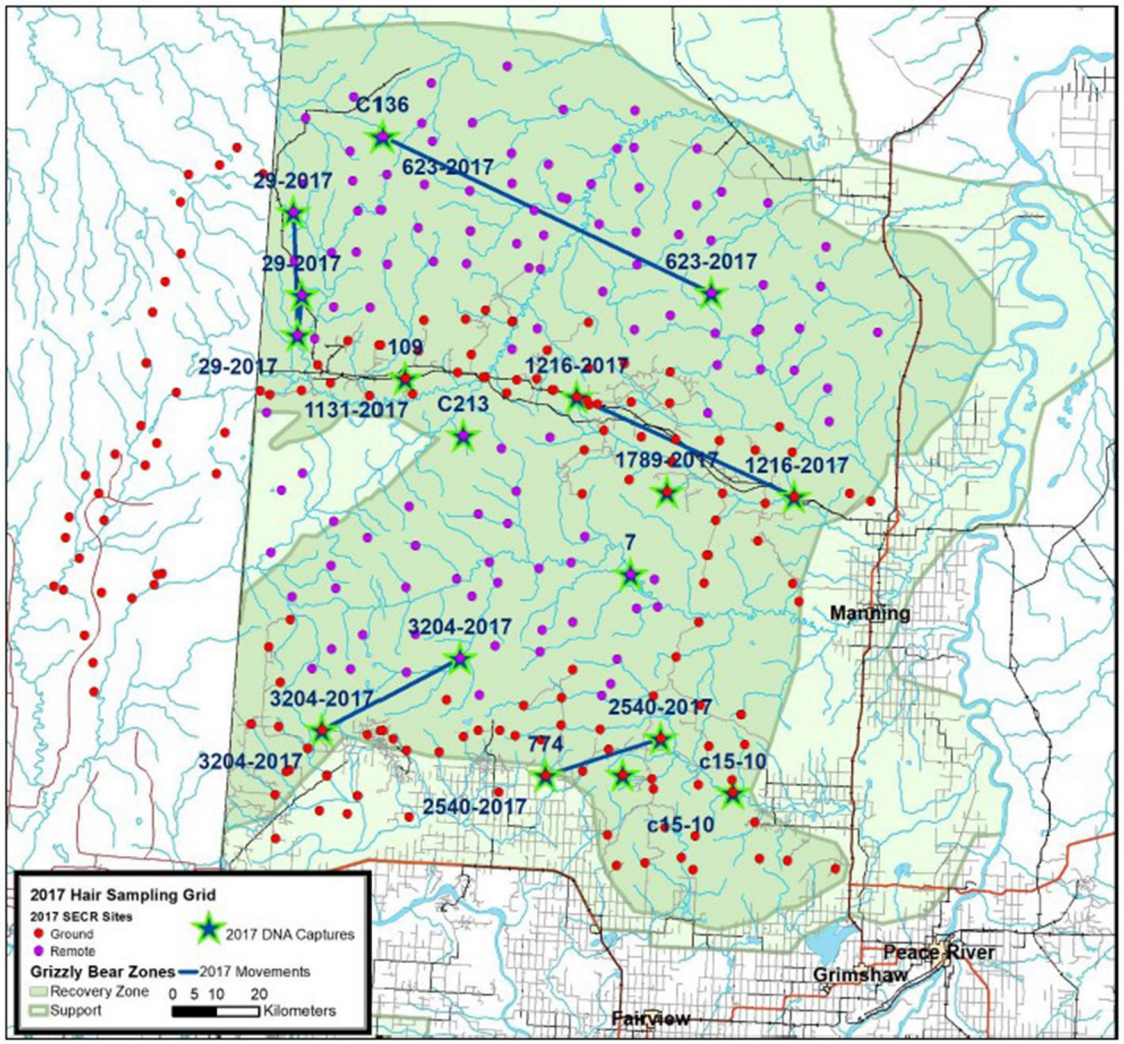


FIGURE 3 | Spatial configuration of the hair snare sampling grid and grizzly bear detections across BMA 1 Recovery Zone and along the British Columbia border. Labels indicate the ID number of detected grizzly bear individuals and associated movements between hair snare sites.

efforts, alongside employing a non-invasive genetic sampling project coupled with spatially explicit capture-recapture models to estimate grizzly bear density (0.70/1,000 km^2) and abundance (16.3 grizzly bears) in the Recovery Zone of BMA 1 in northwest Alberta.

While collaborative governance is a common term used across public administration literature, how it is defined and applied remains vague (Emerson and Nabatchi, 2015). Our study adopted the principle and spirit of a collaborative governance as suggested by Emerson and Nabatchi (2015), which included decision-making processes that constructively engaged people across their different sectors, perspectives, knowledge, experiences and interests to identify the values, needs and objectives for BMA 1 grizzly bear recovery. Our collaborative approach built and nurtured relationships across otherwise disconnected stakeholders and leveraged human and financial capital to conduct the first-ever population estimate for BMA 1 grizzly bears. A major strength in our approach was collectively and iteratively developing the TOR and the TOC, and learning from our collaborative arrangement by evaluating participants' experiences (Emerson and Nabatchi, 2015; Huge and Mukherjee, 2017). Through this, we learned that collaboration provides for opportunities to develop and improve relationships through active participation in discussions, decision-making and, as in our case, supporting data collection and analysis. This is a cornerstone of collaborative governance literature, where the importance of stakeholder engagement in scientific and decision-making processes is increasingly recognized (Redpath et al., 2017). However, we also learned that greater efforts must be taken in future to engage other stakeholders in grizzly bear science and applied management, and that a BMA 1 management plan

TABLE 2 | Grizzly bear density model results, with density modeled as a homogeneous surface in both models.

Model	Density (/1,000 km^2)	RSE	g_0 (e^{-3})	σ	Expected *N*	Δ AICc
$g_0 \sim$ TrapType	0.70 (0.36–1.35)	0.349	6.7 (2.6–17.2)	22,503 (14,776–34,273)	16.3 (8.4–31.8)	0
$g_0 \sim$ TrapType, σ ~ TrapType	0.69 (0.36–1.35)	0.348	4.8 (1.5–15.1)	27,026 (15,057–48,059)	16.3 (8.4–31.6)	13.8

Detection parameters are for models parametrized using meters. Figures presented in brackets represent 95% confidence intervals. Expected population estimates are specific to the BMA 1 Recovery Zone. See DNA Field Methods and Analysis Methods for additional information.

utilizing population estimates and linear footprint results is strongly desired.

Overall, we suggest our efforts to build a strong, collaborative governance structure ultimately helped move grizzly bear conservation and management forward in northwest Alberta (Ansell and Gash, 2008; Stern and Coleman, 2015). By employing a collaborative process that shared the investment of time, money and collective actions, we were able to use the best available scientific techniques to deliver a grizzly bear population estimate for BMA 1. Indeed, the formalization of the NGBT created an open, respectful, and constructive space that made sense for the context in which we were working in and ultimately take steps toward achieving grizzly bear recovery in a multi-use landscape (Redpath et al., 2017).

Specific to grizzly bear population estimation, we found that BMA 1 contains a low density of grizzly bears and is currently the lowest recorded density of an established grizzly bear population in Alberta. Elsewhere in Alberta, grizzly bear density estimates range from 5.25 to 20.4 /1,000 km^2 (Alberta Environment Parks, 2020). We note, however, that our detection probability (g0) for grizzly bears was remarkably low and was substantially lower than the parameters we included within our simulation exercise (**Table 2**, **Supplementary Material B**). Further, our estimates of sigma (i.e., the spatial area over which a grizzly bear can be detected) were very large (**Table 2**). Given this, it is likely that we would need to substantially increase our sampling effort in order to achieve a more reliable (i.e., coefficient of variation <0.2) density estimate. However, increased sampling would be expensive and logistically challenging given the characteristics of this landscape.

We did attempt to increase grizzly bear detections by adding sampling sites opportunistically along existing travel routes (i.e., roads) and including sampling stations in British Columbia. However, our low number of detections precludes us from further examining the influence of these decisions on our density estimates. Indeed, we assumed a uniform density in our SECR models, which is an oversimplification because it is likely that grizzly bear density varies as a function of habitat and proximity to roads (Boulanger and Stenhouse, 2014).

We detected only two female grizzly bears and note that sex-specific demographic forces may be driving a male-skewed sex ratio. It is unclear if this ratio is a true component of this population or an unexplained sampling artifact. Lured rub objects represented only 13% of the sampling sites but provided 39% of grizzly bear detections, with 8 of 20 male detections and 1 of 3 female detections occurring at rub objects. Generally, male grizzly bears rub more frequently than female grizzly bears, though this difference dissipates as the season progresses (Lamb et al., 2016; Morehouse and Boyce, 2016). One factor that may have influenced bear detections and rubbing behavior is that we used lured rub objects, whereas most previously published studies used natural rub objects. It is also possible that more female bears occur outside of the Recovery Zone and therefore did not encounter our traps. One potential explanation for this is that the Recovery Zone represents higher quality habitat and is thus used by the more dominant male bears, with females being excluded or seeking alterative habitat (i.e., dominance hypothesis, Elfstrom et al., 2012; Stewart et al., 2013). This theory remains to be tested and is considered a low probability scenario given the large spatial scale over which this process would be occurring. Regardless, our detected female scarcity elevates the importance of minimizing human-caused grizzly bear mortality and further emphasizes the importance of maintaining female survival in BMA 1 (Alberta Environment Parks, 2020). Educational outreach and access management strategies should therefore continue to be implemented across BMA 1 to mitigate human-bear conflict and the potential for female mortality.

Our project also attempted to measure the transboundary population between Alberta and British Columbia, but no grizzly bears were detected in this portion of our study grid. However, this lack of detections is difficult to link to differences in bear density and should be interpreted with caution (Environmental Reporting BC, 2020). And finally, while we detected few grizzly bears, a large portion of our collected hair samples were black bears, suggesting that our sampling methods were appropriate for detecting bear species. Grizzly and black bears have different life histories, where northwest Alberta's boreal landscape is likely more suitable black bear habitat than grizzly bear habitat (e.g., Bonin et al., 2020).

Despite these challenges, our abundance estimate represents the first empirical estimate for BMA 1 and is a baseline against which future BMA 1 monitoring and management actions can be measured. This includes setting objectives to address the limiting factors (i.e., human-caused mortality) for grizzly bears, implementing local management policy co-designed by the NGBT, exploring opportunities for creative and innovative strategies to implement for this landscape, and assisting in evaluating the efficacy of grizzly bear and human behavior management as well as land use and forestry practices.

Importantly, we also provide a starting point from which to evaluate the achievements and impacts of a complex collaborative governance arrangement for an at-risk species and demonstrate success despite the challenges encountered. Going forward, priority actions the NBGT has considered for local planning includes collectively mapping actual linear footprint and taking coordinated action on motorized (i.e., on-highway vehicle) access and associated impacts (i.e., reduction and restoration of existing linear features, evaluating open road densities and access barriers). Additionally, the NGBT agrees that locally relevant educational outreach, using Alberta BearSmart principles and materials, must be delivered across BMA 1. This includes engagement with petroleum and forest industry, landowners, recreationalists, and community residents to teach and encourage proactive bear safety and conflict mitigation strategies (e.g., electric fencing). Lastly, the NGBT members recognize the need for their organizations' staff and the broader public to contribute to grizzly bear observations, and as such, supports the use of the smartphone-based reporting application GrizzTracker (www.grizztracker.ca) as an effective reporting method.

CONCLUSION

This study represents the first attempt to empirically estimate the grizzly bear density and abundance across Alberta's northern BMA 1, through a multi-stakeholder collaborative arrangement called the Northwest Grizzly Baer Team (NGBT). Collaboration was at the core of our success, which went beyond simply acknowledging the need to collaborate, to enabling proactive participation by NGBT members in project design, implementation and evaluation. In turn, the relationships we fostered enabled us to leverage financial, human, social, and policy resources, and helped to build trust, reciprocity and exchange across stakeholders and the broader public through open and transparent communications (Ostrom, 1990; Pretty and Smith, 2004; Ansell and Gash, 2008; Kallis et al., 2009). Additionally, our collaborative arrangement provided extensive opportunities to co-learn, share, and engage across the broad membership of the Northwest Grizzly Bear Team.

Our summative evaluation was an important step in understanding the efficacy of our collaborative arrangement, helping to provide evidence to inform future decisions on alternative governance structures. Indeed, this team has agreed to continue to build and strengthen relationships, facilitate constructive dialogue, and share data and knowledge going forward, to develop a locally relevant BMA 1 plan. In addition to our collaborative outcomes, our efforts provide a robust grizzly bear population dataset to help fulfill Alberta's recovery policy objectives and can contribute to future performance evaluations of integrated land management. Future collaborative governance arrangements would be well-suited to look to our project as a guideline for establishing multi-sectoral teams that applied various social and natural science methodologies to define and work toward resolving a complex, real-world, species at risk problem. Overall, the relationships, funding, datasets, and commitment to future efforts would not have been possible without coming together in the spirit and practice of collaboration.

DATA AVAILABILITY STATEMENT

The data supporting the conclusions of this article will be made available by the authors upon request.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. The patients/participants provided their written informed consent to participate in this study. Ethical review and approval was not required for the animal study because Animals were not handled. Non-invasive genetic sampling (hair samples) was explicitly used.

AUTHOR CONTRIBUTIONS

CH, LF, LV, and NM conceived of the project. CH and LV directed the project. LF led fieldwork logistics, with participation by NM, CH, and LV and project partners. LV completed the secr analysis, with AM and RS reviewing. CH completed the questionnaire analysis with review by project partners and led the manuscript writing process, with all other authors contributing content and revisions. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

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The First Political-Ecological Database and Its Use in Episode Analysis

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Biodiversity loss is a consequence of socio-ecological processes. Observations on anthropogenic actions toward ecosystems coupled to observations on ecosystem metrics are needed to help understand these processes so that ecosystem management policies can be derived and implemented to curb such destruction. Such data needs to be maintained in searchable data portals. To this end, this article delivers a first-of-its-kind relational database of observations on coupled anthropogenic and ecosystem actions. This Ecosystem Management Actions Taxonomy (EMAT) database is founded on a taxonomy designed to support models of political-ecological processes. *Structured query language* scripts for building and querying these databases are described. The use of *episodes* in the construction of political-ecological theory is also introduced. These are frequently occurring sequences of political-ecological actions. Those episodes that test positive for causality can aid in improving a political-ecological theory by driving modifications to an attendant computational model so that it generates them. Two relational databases of political-ecological actions are described that are built from online news articles and published data on species abundance. The first concerns the management of the East African cheetah (*Acinonyx jubatus*) population, and the second is focused on the management of rhinoceroses (*Ceratotherium simum*) in South Africa. The cheetah database is used to study the political drivers of cheetah habitat loss, and the rhino database is used study the political drivers of rhino poaching. An EMAT database is a fundamental breakthrough because is provides a language for conservation science to identify the objects and phenomena that it is about. Therefore, maintaining political-ecological data in EMAT databases will advance conservation science and consequently, improve management policies that are based on that science.

Keywords: biodiversity loss, cheetah (*Acinonyx jubatus*), rhinoceros, socio-ecological analysis, relational database, ecosystem management, episodes detection, taxonomy

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1. INTRODUCTION

Anthropogenic actions are causing the earth's sixth mass extinction (Ceballos et al., 2015). Such losses occur within *political-ecological processes* wherein sequences of political actions cause social groups to carry out actions that impact ecosystems. Such processes lie at the interface between political science and ecology. The complexity of each of these fields coupled to a pattern of interactions between them can result in highly complex system dynamics. Theoretical understanding of these processes is in its infancy (Bassett and Peimer, 2015). Theory emerges from

efforts to explain observations, and is reinforced or abandoned through the examination of data taken from experiments. Data, then, is critical to the development of theory. As a first step toward structuring data that can support the development of political-ecological theory, a first-of-its-kind database has been developed that is the foundation for creating accessible and searchable databases of political actions that are linked to observations on affected ecosystems. This database is critical to the development of a political-ecological theory of how (a) developed countries interact with developing countries in the management of ecosystems that contain endangered species; (b) groups within developed countries such as conservation-focused *nongovernmental organizations* (NGOs) interact with groups in developing countries such as wildlife management agencies; (c) a country's indigenous people interact with that country's endangered species; and (d) development projects affect a country's endangered species.

Call an ecosystem that contains one or more endangered species, an *at-risk ecosystem*. Although the examples given in this article concern developing countries (Kenya, Tanzania, Uganda, and South Africa), the database developed herein will be of critical importance for understanding the political interactions both within developing and developed countries and between such countries as they affect at-risk ecosystems that span these countries. Specifically, query results from this database can be used to propose and then test a theory of how political actions affect ecosystem management policies; and how the combined effects that these interactions and actions have on what is actually done to conserve or harm an at-risk ecosystem.

Examples of at-risk ecosystems contained in a developed country include the ecosystem of the United States (U.S.) states of Idaho and Montana that contains the reintroduced grey wolf (*Canis lupus*) (Kiasatpour and Whitfield, 2008); and the everglades ecosystem in the U.S. state of Florida that contains the endangered wood stork (*Mycteria americana*). This stork is threatened, in-part, by the invasive Burmese python (*Python molurus bivittatus*) (Dove et al., 2011). And, most challenging, is the Pacific ocean ecosystem that contains the endangered blue whale (*Balaenoptera musculus*) (Haas, 2011, ch. 3). This particular ecosystem spans many developing and developed countries.

Actions are stored in the database in the active voice, e.g., a news article reporting the donation of wildlife monitoring equipment to a wildlife protection agency is entered into the database as a date, actor, and subject-indexed occurrence of the action `donate wildlife monitoring equipment`. Likewise, a news article reporting on the passage of a bill that strengthens wildlife protection laws is entered into the database as a date, actor, and subject-indexed occurrence of the action `strengthen wildlife protection laws`. And a news article reporting on the arrest of members of a wildlife trafficking syndicate is entered into the database as a date, actor, and subject-indexed occurrence of the action `arrest wildlife traffickers`. Entering actions into the database in this way allows it to be queried for particular actions that are taken in a particular time window, by particular actors, and directed toward particular targets (subjects).

This article delivers a multi-faceted breakthrough on how observations on political-ecological actions can be used to develop political-ecological theory. These facets are

1. a first-of-its-kind *relational database* (Churcher, 2016, p. 1–13, 213–231; Coronel and Morris, 2017, p. 72–168; IBM, 2021) to capture the entities that make up political-ecological phenomena, relationships between these entities, and the attributes of these entities;
2. *structured query language* (SQL) scripts (Coronel and Morris, 2017, p. 246–415) for implementing this database technology;
3. the first application of *episode* detection to query results from a political-ecological database; and
4. exemplary use of this relational database to assess political action correlates of east African cheetah habitat loss, and political actions affecting rhino poaching in South Africa.

A political-ecological process or *system* is also called a *socio-ecological* system or a *social-ecological* system (e.g., see Virapongse et al., 2016). The former term is used herein for the following reasons. Ecosystems respond to actions. Hence, the relevant social systems for understanding anthropogenic effects on biodiversity are those that produce actions that affect ecosystems. Political systems are social systems that produce ecosystem management policies and have the capability of using force as necessary to assure these policies are implemented (Moe, 2005; Barthwal and Sah, 2008). In other words, the key characteristic of a political system is that it is capable of exercising coercion. Several groups influence a political system including (a) the country's executive, legislature, and judiciary; (b) international NGOs; (c) organized crime syndicates; and (d) indigenous peoples, racial minorities, and religious minorities. Of these, only group (a) can yield formal governmental power.

But any social system, political or otherwise, is made up of individuals. And many individuals are driven by the need for power (Guinote, 2017). These power-seeking individuals engage in political actions that they believe will increase their sense of power. They do this because political systems can, once controlled, deliver manifestations of the power that these individuals seek. Ultimately, these manifestations of power impact an ecosystem. Hence, a political-ecological system is a complete and precise way to describe the causal chain that starts with individuals wanting power, through political systems that they use to get it, and ultimately to ecosystems that are impacted by its manifestations. On the other hand, the phrase "socio-ecological system" does not convey this power-driven chain of actions.

Data on intentional ecosystem management actions (such as creating a wildlife reserve), and unintentional ones (such as poaching) that is coupled to data on associated ecosystem metrics needs to be easier to use. Doing so would make it easier to assess the effects of management policies on targeted ecosystems (U.S. National Science Foundation, 2014). Rissman and Gillon (2017) believe that links between ecological and social dynamics including feedback loops are needed to inform policy and management and improve both social acceptance and ecological effectiveness of conservation strategies. This view is echoed by Bodin et al. (2014). As Laurila-Pant et al. (2015) develop

a framework for incorporating biodiversity protection into ecosystem management policies, they see a need for large, multi-disciplinary data sets that contain responses to environmental policies and the costs of those policies. And, Leenhardt et al. (2015) see a lack of standardized data on social-ecological systems that links changes in ecological processes to social responses—including attendant feedback loops. These authors believe this scarcity of social-ecological data is limiting the development and validation of social-ecological models.

In response to this general plea for more data structure, Frey and Cox (2015) call for ontologies of political-ecological phenomena to be created.These authors see the building of political-ecological theory being hampered by a lack of data compatibility across different theory-building efforts, and note that data sets collected by different research teams are rarely shared or integrated. They see this data incompatibility challenge as contributing to what they call a "scatter problem:" "...a lack of integration of many research findings into a cohesive set of theoretical instruments that explain how relevant conditions interact to produce success or failure over time." These authors propose the use of ontologies to help reveal observations that are related between different political-ecological databases. Using such data structures would allow integration of different theory building efforts into a comprehensive theory of how politics affects and is affected by the environment. These authors also argue that an ontology supports formal (e.g., relational) database queries via searches based on pre-defined attributes and hence is the preferred way to organize political-ecological data because it structures, unifies, and formalizes the represented knowledge. These characteristics would also allow such knowledge to be reused.

Therefore, an accessible and searchable database of political-ecological actions is seen as a necessary precursor for developing a theory of political-ecological systems and for developing tools to manage such systems. To aid these two development agendas, an *ecosystem management tool* (EMT) has been developed (Haas, 2011, p. 59–78; Haas, 2021) that includes tools for organizing political-ecological data into a relational database. This database implements the *ecosystem management actions taxonomy* (EMAT) of Haas (2011), p. 123–141 and Haas (2018). This taxonomy classifies and indexes political-ecological actions.

A taxonomy is an ontology that represents only hierarchical relationships among its members (American Society for Indexing, 2018). Whereas, in an ontology, there are more types of relationships and these relationships are more specific in their function. Further, information conveyed through indexing in a taxonomy is embedded into the ontology itself (American Society for Indexing, 2018). Taxonomies are also known as *hierarchical ontologies* (Khan and Safyan, 2014). The McKenna/Bell classification system for mammals (Wilson and Reeder, 2005) is an example of a taxonomy. This taxonomy's hierarchy is *Subclass, Infraclass, Supercohort, Cohort, Magnorder, Grandorder, Order,* and *Mirorder*. Another example is the *enterprise ontology* of Dietz (2006). This taxonomy structures those actors and processes that constitute the functioning of an enterprise, e.g., a manufacturing firm.

A *relational database management system* (RDBMS) is the computer system (hardware and software) needed to host the database itself (Mata-Toledo and Cushman, 2000, p. 1). The software component is used to build the database and query it. The present article focuses on the database and software components of a RDBMS. Queries against a relational database are often expressed in SQL. SQL is well-established and used to query about 80% of all databases currently in existence (DB-Engines, 2017). Call a relational database of political-ecological actions developed herein, an *EMAT database*, and the computer system that serves an EMAT database, an *EMAT RDBMS*. Let an observed political action or an observation on an ecosystem metric that has been matched to a member of the EMAT be referred to as an *EMAT action observation*.

A brief tutorial on relational databases appears in **Appendix A**.

Once built, queries against an EMAT database can be used to

1. extract ecosystem actions of a selected type, date range, and country for purposes of assessing an ecosystem's sustainability;
2. form political-ecological data sets to build political-ecological theory through the use of *episode analysis* (described below);
3. construct a data set of EMAT action observations in order to statistically fit the parameters of a simulation model of a political-ecological system (Haas, 2011, p. 161–178);
4. help construct and evaluate ecosystem management policies; and
5. critique, modify, and/or extend the EMAT.

This article delivers as its central contribution, the needed foundation for structuring and using political-ecological data: an EMAT database. Then, as an example of one of the many uses of an EMAT database, episode analysis is developed and used in order to show how it can give insight into a political-ecological system.

Shneider (2009) describes four stages that a new science such as conservation, goes through as it develops. These are (I) the identification of its fundamental objects, phenomena, and language to describe its subject matter; (II) creation of tools for studying these objects and phenomena; (III) discovery of mechanisms that predict observed phenomena; and (IV) broadcast and maintenance of this predictive knowledge. This article delivers a taxonomically-based relational database of political-ecological physical actions, verbal actions, and data. It further develops the concept of an episode of political-ecological actions, and gives a tool for determining if such an episode is causal. These two breakthroughs: an EMAT database and attendant episode analysis give, for the first time, a language to conservation science. This language enables researchers to identify what data needs to be collected, and what a theory of political-ecological systems should be able to explain. In particular, this theory should offer data-verified causal mechanisms that produce the observed, coupled actions of political actors and ecosystem members. This operational triad of objects, phenomena, and language supports the convergence of conservation theories. Therefore, this article makes a fundamental contribution to conservation science

because it completes stage I of a developing science through its EMAT database, and begins stage II through its introduction of episode analysis.

2. MATERIALS AND METHODS

2.1. EMAT Database Overview

On the political side, an EMAT database holds ecosystem-affecting anthropogenic actions as reported in news articles (hereafter called *stories*). Many of these stories are available online. The story's *source* is the news outlet responsible for the story, e.g., The Huffington Post. Some of these actions can be matched to members of the EMAT. On the ecosystem side, an EMAT database holds actions taken by the ecosystem, and references to ecosystem data sets rather than the data sets themselves. Such data sets are modeled as being observations on the EMAT action `collect data`. This action refers to a set of observations on an ecosystem metric.

Recent efforts to collect biodiversity data have produced large ecological datasets such as the Global Biodiversity Information Facility (GBIF), eBird, iDigBio, and iNaturalist (Heberling et al., 2021). Access to these strictly ecological datasets is free and hence selected subsets of them would be easily added to an EMAT database through the `collect data` EMAT action. Before these datasets can be accessed, however, spatial location of sensitive species observations needs to be generalized so that poachers cannot use the resulting dataset to locate species. There have been several efforts aimed at dealing with this problem (Haas, 2018; Chapman, 2020) although it remains possible (but not likely) that a determined poaching syndicate could hire analysts to reverse engineer species locations.

2.2. The EMAT

See Haas (2018) for a detailed description of the EMAT. In brief, the EMAT consists of 632 physical and verbal actions broken into five categories: military, diplomatic, economic, environment, and ecosystem. The fourth category consists of anthropogenic actions directed at the environment, e.g., `clear new land` or `collect data`. The fifth category consists of actions taken by non-anthropogenic actors, e.g., (elephants) `trample crops`—a frequent occurrence in parts of East Africa.

Many of the actions in this taxonomy have been parsed into three *equivalence sets*: A set of semantically equivalent *m*-word verbs, a set of semantically equivalent direct object phrases, and a set of semantically equivalent prepositional phrases. Letting *m* be a positive integer, an *m*-word verb subsumes single-word verbs (either *regular* or *irregular*), and *multi-word* verbs (those that use more than one word to convey their meaning, e.g., "find out") (British Council, 2017). See Aarts (2011) for definitions of direct object phrases, and prepositional phrases.

The EMAT, being an ontology of socio-ecological actions, is an exemplar of what Frey and Cox (2015) see as being needed to advance socio-ecological theory.

2.3. Data Acquisition Is Performed Outside of the Database

The task of extracting actions from stories is a cognitive/linguistic data processing activity, namely, shallow parsing coupled to phrase similarity computations. These tasks have little to do with organizing, linking, or querying an existing set of observed political-ecological actions. Therefore, data acquisition is kept separate from an EMAT database. Indeed, there are advantages to separating data acquisition from the building and querying of a relational database. These include the following.

1. Different software systems running on possibly different hardware at different locations can be used to acquire data without the need to transfer such systems to a central location and translate them into a single database language (Nielsen et al., 2013).
2. Computationally intensive data acquisition schemes can be run on special-purpose high performance computing systems without the overhead of an overarching RDBMS.
3. Standard, open source relational database software such as MySQL (Grippa and Kuzmichev, 2021) or Java DB (DB-Engines, 2018) can be used without modification to build and query the database.

See Biermann (2014) for a web-based system that follows this approach of keeping data acquisition separate from database creation, and database querying.

Only sentence components are stored in an EMAT database—not the sentence text itself. This design decision keeps the natural language parsing step (called here, the *parsing preprocessor*) separate from the steps of EMAT database creation and EMAT database querying.

2.4. Design of an EMAT Database

The design of a relational database begins with the definition of entities and the relationships between them (Coronel and Morris, 2017, p. 117–168). EMAT database entities are stories, sentences, noun phrases, *m*-word verbs, direct object phrases, prepositional phrases, and EMAT actions. The database schema is hierarchical with stories being at the lowest level followed by sentences; followed by noun phrases, *m*-word verbs, direct object phrases and prepositional phrases—these latter four being at the same level. The highest level in the hierarchy holds the EMAT actions themselves. Actions toward an ecosystem such as `open a wildlife reserve to settlement` are particular EMAT actions. Reactions by the ecosystem to these anthropogenic actions are also EMAT actions. Examples of the latter include the EMAT actions of `trample crops`, and values on landuse over a region contained in a `collect data` EMAT action.

An *entity relationship diagram* illustrates this design (**Figure 1**) wherein for example, an *m*-word verb can map to many sentences and a sentence can map to more than one *m*-word verb: a many-to-many relationship. The requisite junction table in this case is the Figure's `mwvrbsen` table.

The name of the group responsible for an action is contained in that observation's noun phrase entity. The parsing

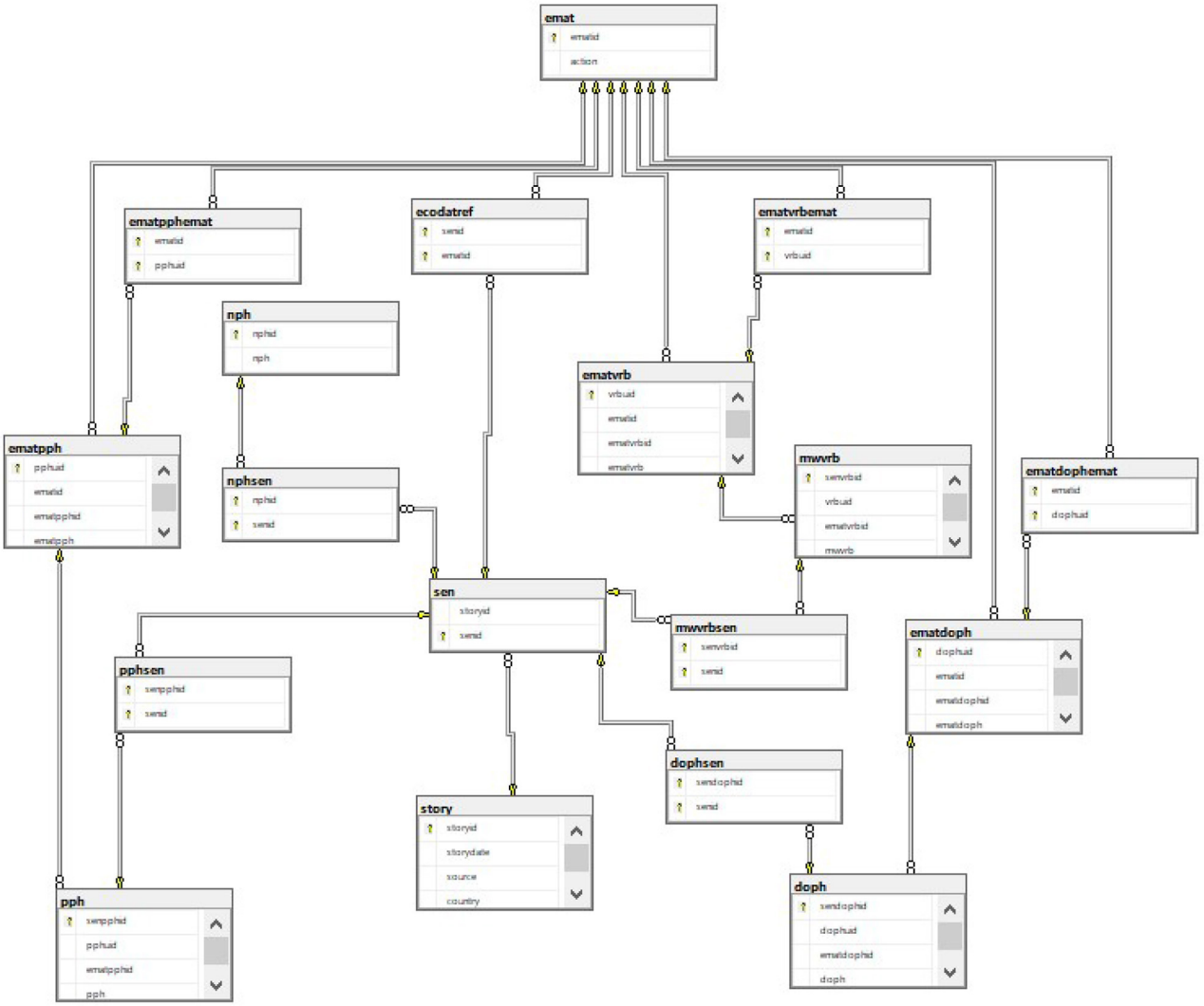


FIGURE 1 | Entity relationship diagram of the EMT database drawn using the Database Diagram™ tool in SSMS. Rectangles are entities. Rows within rectangles are attributes that take on values. An arrow into an entity indicates a source entity can map to only one entity whereas a line indicates a source entity can map to many entities.

preprocessor has assigned this value. Hence, noun phrase entities are exclusively group names, e.g., *Kenya Wildlife Services*, or *rhino poachers*. A group is akin to the social object of an organization as highlighted by Hanneman and Shelton (2011).

Entities that model political action observations have been designed to be at a finer scale than EMAT actions so that new EMAT actions can be added to the EMAT by running learning algorithms that discover new EMAT actions. These algorithms do this by querying an EMAT database for new combinations of group names, *m*-word verbs, direct object phrases, and prepositional phrases (Haas, 2021, **Appendix**). This approach of building a database around parts-of-speech to allow unforseen entity combinations to be discovered is similar to the approach taken by Davies (2005) in his development of a database of the Spanish language.

Entities in the *data set reference* table (table `ecodatref` in **Figure 1**) are observations on the `collect data` EMAT action and have seven attributes: `source`, `species`, `type`, `country`, `region`, `startdate`, and `enddate`. The `source` attribute is either *observation*, or *model*. The `species` attribute indicates the observed species, e.g., *cheetah, rhino,* or *cycad*. The `type` attribute can take on the values of *abundance, presence/absence, capture-recapture, rainfall, NDVI*, and *landuse*. For these latter three values, the species attribute is set to *N/A*. These data set references are preprocessed into one-sentence stories of the form "*group name* collected *type* data on *species* in *region, country* during the period *startdate* to *enddate*." *group name* is the group who collected data, e.g., *Kenya Wildlife Services*, *SANParks Scientific Services*, or *TerraServer*™.

2.5. Software Implementation

The EMAT RDBMS is implemented as an *embedded* Java DB™ database (O'Conner, 2006) within the author's **id** software system (see **Appendix B**). This system also contains the parsing preprocessor. An EMAT database is built and queried via **id**'s `rdbms_()` relation. This relation's syntax is

`rdbms_(`*database_name groups_file_name regions_file_name parsed_stories_file_name*
line_1 . . . line_m$_1$ endcommand
[*line_1 . . . line_m$_2$ endcommand*]
⋮
[*line_1 . . . line_m$_n$ endcommand*]
`endquery` *option*)

where *line_1 . . . line_m$_i$* is an m_i-line SQL query, and *option* is one of `build`, `update`, or `use_existing_database`.

An EMAT database as described in section 2.4 is created when the `build` option is set. This task entails loading the EMAT and each story's sentence components (m-word verbs, direct object phrases, and prepositional phrases) into corresponding tables of the Java DB database.

Extracting an EMAT action from a sentence in a story and storing it in an EMAT database is a two-step procedure. The first step, that of retaining a sentence component only if its similarity score is >0.95, is performed within the parsing preprocessor (Haas, 2021, **Appendix**). For a given set of sentence components, the second step consists of computing α, the sum of the m-word verb similarity score, direct object phrase similarity score, and prepositional phrase similarity score. Then, the associated EMAT action is entered into the database only if α is greater than a threshold value. Some sentences that contain EMAT actions may have no prepositional phrase. For these sentences, α cannot be larger than 1.9. This fact motivates setting the threshold to 1.9 for all EMAT databases reported herein.

2.6. Episode Discovery and Causality Testing

An actions history produced by an SQL query against an EMAT database can be used in many ways to aid the development of political-ecological theory. One such way is to characterize the dynamics of political actions by finding repeating temporal patterns of actions, i.e., sequences of actions. Such sequences are called *frequent episodes* in computer science (Ma et al., 2004). Episodes may give some idea of how a group responds to actions of other groups or the ecosystem.

Episodes may also indicate system behaviors that computational theories of political-ecological systems should be able to reproduce. Indeed, a frequently-occurring episode in an observed actions history may be the result of a causal relationship among the groups and ecosystem generating those actions. The presence of such an episode should lead the researcher to apply a test for Granger causality (Budhathoki and Vreeken, 2018) and then examine the political-ecological model for its ability to reproduce the episode should it pass such a test. Call this two-step activity of first computing episodes, and then subjecting them to statistical tests for causality, *episode analysis*.

Python code for conducting the "CUTE" statistical hypothesis test of whether one EMAT action is causing another EMAT action (two time series of binary actions) is available from Budhathoki and Vreeken (2018). To conduct this test, one would first query an EMAT database for only the two actions in question, and then compute the test statistic from the query results. Before the causality of episodes involving more than two EMAT actions can be tested, the CUTE statistic needs to be extended. This extension is straightforward according to Budhathoki and Vreeken (2018).

Observed time series of political-ecological actions is a type of an *observational study* (Rosenbaum, 2002) and hence does not meet the assumptions for a randomized controlled experiment. Granger causality, however, is a widely accepted definition for the effect that one time series has on another. The *compression-based identity test* operationalized by the CUTE statistic provides a means for determining if one sequence of actions is causing another sequence of actions in the sense of Granger causality.

2.7. Constructing Complex SQL Queries

Building complex SQL queries to run against an EMAT database can be challenging. Aids for this task exist. For instance, *visual query builders* can help researchers construct complex queries through a graphical user interface that does not require knowledge of SQL. One such tool is the *Query and View Designer*™ that is part of Microsoft's *SQL Server Management Studio*™ (SSMS) (Microsoft, 2017). Zhang and Yi (1998) and Pankowski (2017) give strategies for developing complex queries.

3. RESULTS

3.1. Example 2

Three online stories about rhino poaching in South Africa (see **Figures C1–C3**, **Appendix C**) are used to build a database and from that, identify any EMAT action observations they may hold. First, a parsing relation is run in **id** on the file `dbex.txt` to produce a file of parsed stories, `parseddbex.txt`. Then, the desired database query is executed in **id** via the `rdbms_()` relation shown in **Figure 2**. The database is named `polecol`, and is built from the parsed stories contained in the file `parseddbex.txt` using group names contained in the file `sarhinogroups.dat`, and region names contained in the file `sarhinorgns.dat`. This run produces a list of EMAT action observations (**Table 1**).

The SQL query of **Figure 2** consists of selecting only unique (`distinct`) entries from a list formed by joining records from the m-word verb table and the direct object phrase table that match on their story source, sentence source, and EMAT action attributes. This complex query appears simple when visualized (**Figure 3**).

Figure 2 is central to this article's main point: The **id** language relation shown therein is the only interface to an EMAT database that a researcher needs for purposes of extracting data. The idea is to embed the political-ecological system being observed into one software system, here, **id**, along with the formal database language, SQL to run queries against it.

```
report prepare_data
   rdbms(polecol sarhinogroups.dat sarhinorgns.dat parseddbex.txt
      select distinct v.storyid, v.senid, v.mwvrb, d.doph, v.action
      /* List all EMAT actions found in the parsed sentences of dbex.txt. */
      from (
         /* mwvrb subquery. */
         select subvs.storyid, subvs.senid, subv.mwvrb, action
         from (select * from mwvrb, ematvrbemat
            join emat on emat.ematid = ematvrbemat.ematid
            join ematvrb on ematvrb.vrbuid = ematvrbemat.vrbuid
            where mwvrb.mwvrb = ematvrb.ematvrb) subv
         join (select sen.storyid, mwvrbsen.senid, mwvrb from mwvrbsen
            join mwvrb on mwvrb.senvrbid = mwvrbsen.senvrbid
            join sen on sen.senid = mwvrbsen.senid) subvs
         on subv.mwvrb = subvs.mwvrb) v
      join (
         /* doph subquery. */
         select subds.storyid, subds.senid, subd.doph, action
         from (select * from doph, ematdophemat
            join emat on emat.ematid = ematdophemat.ematid
            join ematdoph on ematdoph.dophuid = ematdophemat.dophuid
            where doph.doph = ematdoph.ematdoph) subd
         join (select sen.storyid, dophsen.senid, doph from dophsen
            join doph on doph.sendophid = dophsen.sendophid
            join sen on sen.senid = dophsen.senid) subds
         on subd.doph = subds.doph) d
      on v.storyid = d.storyid and v.senid = d.senid and v.action = d.action
      endquery
      build)
```

FIGURE 2 | An `rdbms_()` relation to build the `polecol` EMAT database of Example 2 and query it for EMAT actions.

TABLE 1 | Sentence components found in Example 2's three stories by the EMAT action extraction algorithm along with associated EMAT action observations.

Story and sentence	*m*-word verb	Direct object phrase	Prepositional phrase	Observed EMAT action
1, 1	Discovery	Of dead rhinos		Sell a few rhino horns
1, 2	Arrests	Of alleged poachers		Arrest rhino poachers
1, 12	Intercepted	Consignment	Of rhino horn	Seize rhino horn
2, 18	Combats	Rhino poaching	With new regional Court	Increase antipoaching Enforcement
3, 25	Shot dead	Rhino		Sell a few rhino horns

3.2. Two Operational EMAT Databases

Two EMAT databases capable of supporting theory development have been constructed. The first focuses on actions that affect the cheetah population in East Africa (Kenya, Tanzania, and Uganda) (Haas, 2019a). The second focuses on the rhino population in South Africa (Haas, 2019b). Those stories pertaining to East African cheetah that were successfully parsed are contained in the database named `east-af-cheetah`. Likewise, those stories pertaining to South African rhinos that were successfully parsed are contained in the database named `south-af-rhinos`. The set of raw HTML files that contain these stories along with statistics that measure the EMAT action extraction algorithm's productivity are shown in **Table 2**.

These two applications have been chosen in order to highlight the ability of the EMAT database technology described herein to capture fundamentally different political mechanisms and actions that affect and are affected by the actions of an at-risk ecosystem.

3.2.1. Action Extraction

Figures 4, **5** show the time series of the actions extracted by the EMAT action extraction algorithm of section 2.5 applied to the East African stories, and South African stories, respectively (see **Table 3**). Cheetah abundance data is from Durant et al. (2017), IUCN/SSC (2007), and TMAP (2008). Note the prominence of reports of human-wildlife conflict in the East African press, and the prevalence of rhino poaching reports (`sell a few rhino horns`) in the South African press. These two Figures highlight the dynamic nature of this data and also the temporal linkages between anthropogenic actions and ecosystem reactions to them.

TABLE 2 | Raw HTML story files available on the East African cheetah ecosystem management tool website (Haas, 2019a), and South African rhino ecosystem management tool website (Haas, 2019b).

Phrase used for web search	File name	Number of online stories	Number of stories parsed
Stories associated with East African cheetah			
	Efstories-manual.txt	605	279
	Efstories-alerts.txt	1,526	359
"Cheetah"	Efstories-script09-13.txt	695	243
	EFSTORIES-script13-14.txt	3,101	1,438
	Efstories-script14-15.txt	1,856	1,130
	Efstories-script15-16.txt	6,519	2,053
	Efstories-alerts16-18.txt	3,519	1,339
	Efstories-alerts18-19.txt	3,644	958
Totals		21,465	7,799
Stories associated with South African rhinos			
	Kruger080313.txt	1,184	452
"Kruger rhino poaching"	Kruger103114.txt	986	265
	Kruger120715.txt	1,081	339
	Kruger5417.txt	473	123
Totals		3,724	1,179

Stories are scraped from online news aggregators using the given search phrase. An article is successfully parsed if the algorithm can detect a date, country, and at least one action. Performance measures on the number of stories successfully parsed demonstrate the parsing preprocessor's productivity.

3.2.2. Episodes

Because an episode is a sequence of actions, each action in a particular episode has a *position*. The Position Pair Set (PPS) algorithm given in Ma et al. (2004) is used to discover such episodes in the `east-af-cheetah` database (**Table 3**). The first East African cheetah episode is reflective of the attention that human-wildlife conflict receives in the East African press. The first episode in the South African rhino actions history reflects the nearly constant repetition of rhino poaching reports in the South African press.

3.2.3. Cheetah Habitat Destruction

The East African cheetah EMAT database can aid research into the political antecedents that drive reductions in cheetah habitat. Ideal habitat for cheetah is nearly open plains with a shrub cover of about 40%, and a few kopjes (rocky outcroppings) (Broekhuis, 2017). Such habitat can be lost through several politically driven or sanctioned actions including the degazzetting of wildlife reserve land, clearing land for farms or ranches, urban sprawl, and the construction of new roads (Learn, 2017). The four SQL commands and one SQL query used to extract all political actions that negatively impact cheetah habitat are shown in **Figure 6**. The resulting actions history appears in **Figure 7**.

Figure 6 is complex SQL query that is composed of two separate SQL commands: The creation of two temporary tables (`v` and `d`) followed by a `join` operation on them. Temporary table `v` collects *m*-word verbs associated with cheetah habitat-damaging political actions, and temporary table `d` collects those direct object phrases that are associated with these actions. Entries in these two tables are then selected that match on their story source, sentence source, and action attributes.

There are 11,620 stories in the `east-af-cheetah` database. This SQL query requires 45 min on a PC running at 3.0 GHz. As can be seen by the plot, these actions are regular and on-going. Two frequently-occurring episodes that involve the habitat-destroying action `open reserve to settlement` are shown in **Table 4**. These episodes suggest that this action may be a consequence of the decision to invest in tourism but an antecedent to wildlife crime. Having discovered these temporal associations through episode extraction, statistical testing might reveal that such sequences are actually causal.

4. DISCUSSION

4.1. Related Work

4.1.1. Relational Databases of Socio-Ecological Data

A search was conducted of the peer-reviewed literature for reports on the development of relational databases of temporally-indexed observations on social actions that are linked to observations on ecosystem metrics. None were found. Such data is in contrast to databases of case studies of social-ecological systems such as SESMAD (Cox, 2014). What was found, however, were three non-relational data sets that are somewhat similar to what was sought. In the most relevant of these (Xie et al., 2019), the authors construct a socio-ecological data set by coupling socioeconomic statistics to climate data. Then, they use their data set to discover feedback loops between grassland productivity and human actions. Two less-similar data sets are (a) a data set of one-time observations on social and ecological variables within the Amazon basin (Lima et al., 2016); and (b) a one-time data set of descriptive (non-event) social-ecological data on pastoral systems in Mongolia (Laituri et al., 2015).

4.1.2. Construction of Social Theory

Davies (2005) describes a landmark relational database of a corpora of medieval and modern Spanish texts. This author argues that the relational database structure allows investigators to query the database for output to test new theories about how that language evolved. In the field of law, Ribary (2020) presents a relational database of Roman law and argues that queries against it will help tie together many investigations into how legal systems in the ancient world worked. And in the related field of criminal justice, there are several relational databases of terrorists and terrorism events (Bowie, 2017).

The potential for relational databases to aid the development of social science theory in general is discussed in Hanneman and Shelton (2011). These authors address the question of how one would mine several different social process databases including (a) the periodical literature, (b) business directories, and (c) international trade digests. These authors see such potential because of the critical role in social science theory development already played by the databases *Sociological Abstracts* (ProQuest, 2010), and *Web of Science* [Institute for Scientific Information (ISI), 2010]. Similarities are drawn between social processes and the database concepts of an object (entity), and relations

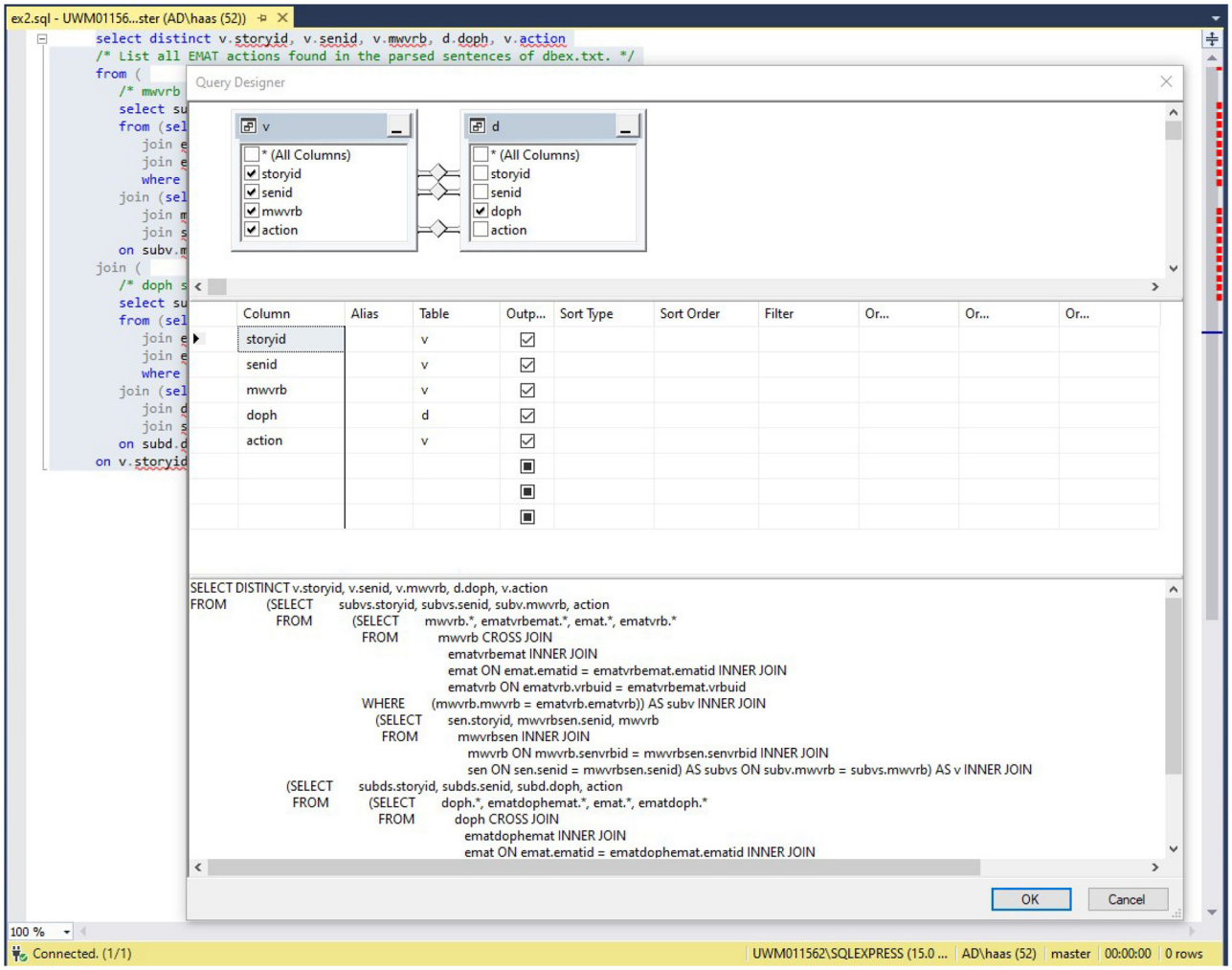


FIGURE 3 | Query diagram of the SQL query of **Figure 2**. The diagram appears in the *Diagram pane* of the Query and View Designer tool. This pane allows query design through drag-and-drop mouse operations. The middle screen is the *Criteria pane* and allows query design through spreadsheet-type entries. The bottom screen is the *SQL pane* and contains the parsed version of the hand-written SQL code that appears in the background window. The red bars on the right-hand side of this window indicate the beginning and content of SQL `select` commands.

between entities. A distinction is made between social objects that are not people such as events and organizations that do, however, possess attributes and agency. These authors point out that the development of social theory is aided when these objects can be classified by their similarities, i.e., similar values on their relational database attributes. And, they argue that social transactions can be modeled as entities connected through relational database linkages. Doing so would enable theories of information flow between social groups to be postulated and tested.

One could conclude from these remarks that building a relational database of a temporal, dynamic social process can contribute to the understanding of that process's dynamics. And further, having such a database would allow the use of the powerful analytic tool, SQL to extract theory-motivated subsets of data from it.

4.2. Peripheral Work

4.2.1. Flat Files?

Why not use a simple collection of text-based computer files to hold political-ecological data? Managing data in this manner is commonly referred to as maintaining a *flat file database* (Database Management System, 2017). If built within a relational database software system, a flat file database is a single-table relational database. If instead, the database is implemented as a set of computer files outside of a relational database software system, the resultant database will suffer from several deficiencies as follows.

1. There is no data model. Hence, a common ontology of political-ecological actions cannot be developed and shared among researchers.

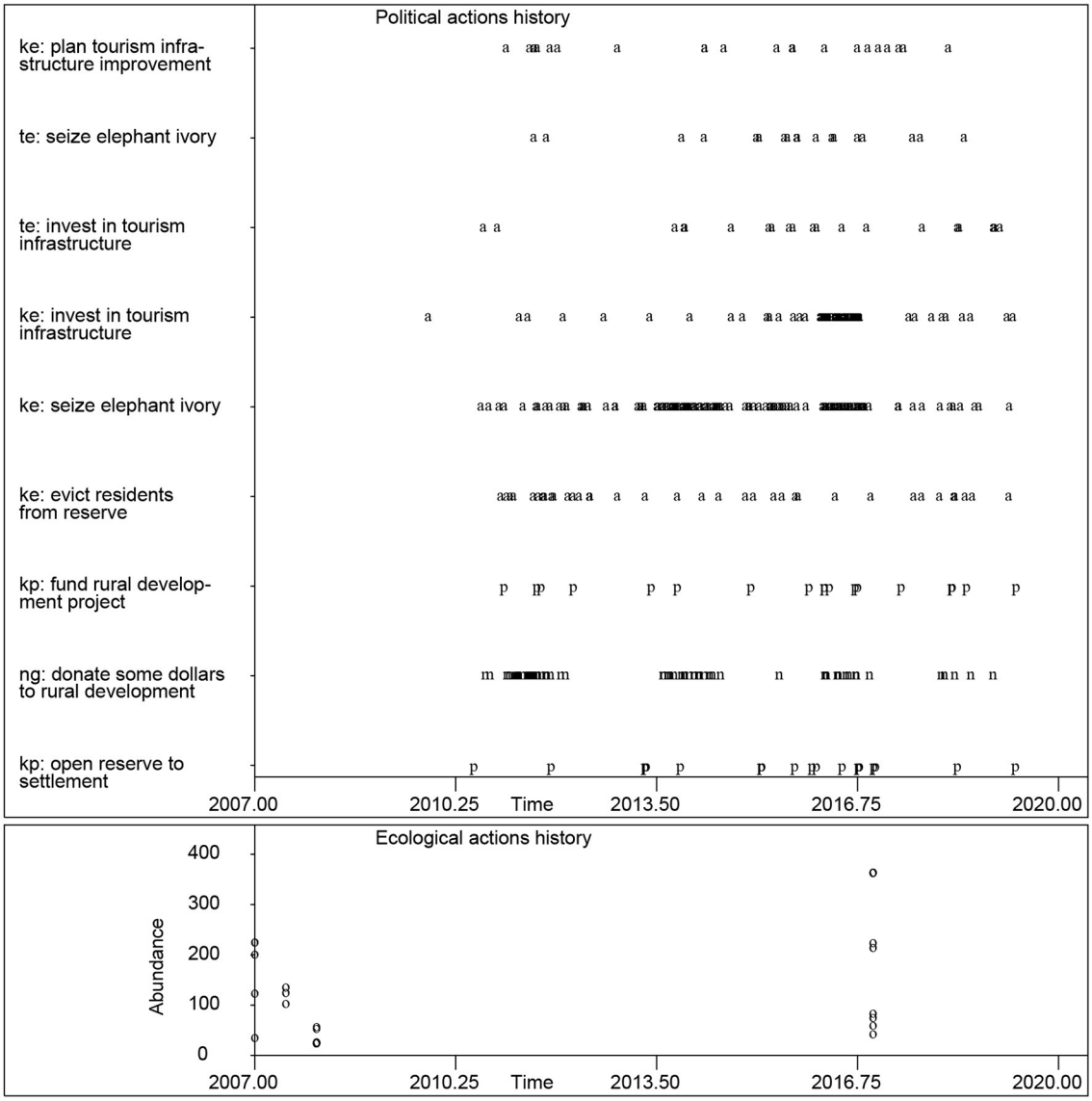


FIGURE 4 | Observed actions history from East African online news stories for the period from January 2007 to June 2019. The presidential office of Kenya, Tanzania, and Uganda are designated by `kp`, `tp`, and `up`, respectively. Similarly, the environmental/wildlife protection agencies are designated with `ke`, `te`, and `ue`; non-pastoralist, rural residents with `kr`, `tr`, and `ur`; and pastoralists with `ka`, `ta`, and `ua`. The group of conservation NGOs who have operations in at least one of these countries is represented by `ng`. The plotting symbol `p` indicates an action taken by a presidential office, `a` an action taken by an EPA, `r` an action taken by rural residents, `s` an action taken by pastoralists, `d` an action taken by a developer, `s` an action taken by pastoralists, `t` an action taken by tourists, `l` an action taken by a legislature, `j` an action taken by a judge, and `n` an action taken by an NGO. Only frequent out-combinations are shown. The bottom plot is observed cheetah abundance.

2. Only pre-programmed queries can be processed. This is called *program-data dependency*. Unanticipated queries (*ad-hoc* queries) can only be made by writing and then executing custom application programs. But, because ecosystem management is an emerging discipline, it is not possible to know in advance what forms of political-ecological data that modelers and policymakers will require. Hence, *ad-hoc* queries will be the norm rather than the exception. In addition, users of a flat file database need to be skilled in writing computer programs. In light of the above-mentioned need for the development of ecosystem management tools aimed at creating sustainable ecosystems, researchers, and policymakers from a wider range of backgrounds will need to become involved. Only a subset of these individuals can be expected to have such requisite programming skills. A lack of such skills then, has the potential of making data access a critical bottleneck to more effective management of at-risk ecosystems.

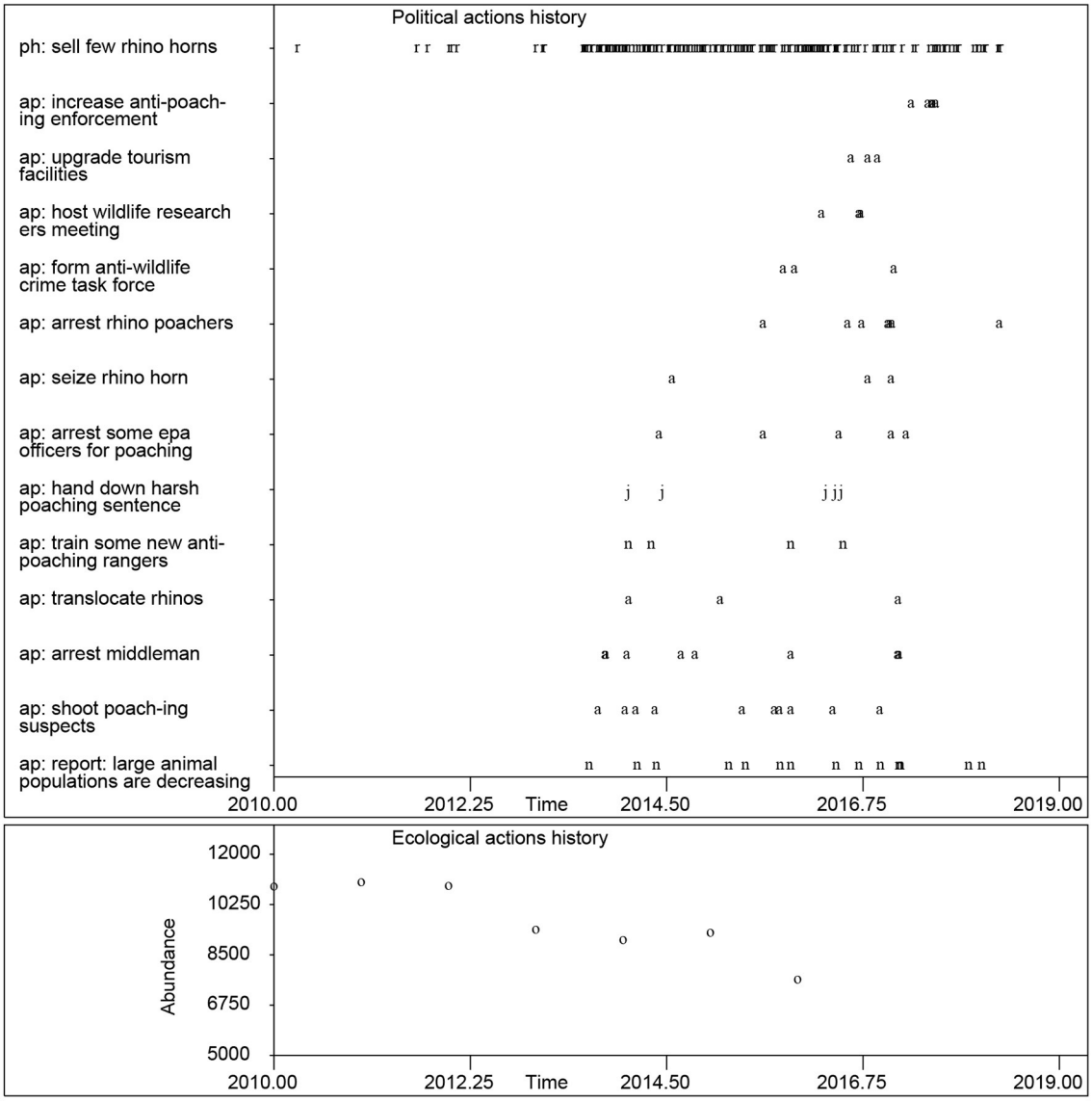


FIGURE 5 | Observed actions history from South African online news stories for the period January 2010–June 2019. Rhino poachers are designated by `ph`. South African rangers and administrators engaged in anti-poaching activities are designated by `ap`. Rhino abundance data is from Haas and Ferreira (2018). See **Figure 4** for the plotting symbol legend.

3. Data isolation can occur when distinct collections of data that are instances of the same entities are held in different files as there is no mechanism to recognize such relationships.
4. If data on two or more entities is redundant, data inconsistencies can emerge. But a flat file system has no mechanism to guard against such redundant data.
5. Data corruption can occur through multiple users accessing the same records of one or more of the files. These events are called *concurrent access anomalies*.
6. No checks are made on data integrity, e.g., a restriction on the range of an attribute.
7. No security protocols are implemented to control who has access to the data.

See CareerBless (2018) for further details of these deficiencies. Wesley (2000) gives a step-by-step tutorial on how a flat file database can be improved by migrating it to a relational database.

4.2.2. Merging Heterogeneous Databases

A topic related to schema development is the merging of heterogeneous databases (e.g., Karasneh et al., 2009). These authors introduce a *relations schemas matcher* algorithm that produces a measure of similarity between the names of tables in two different databases. This syntactic phrase similarity measure is related to the *SIM(phrase_1, phrase_2)* algorithm given in Haas (2021) (**Appendix**). The focus of the present work, however, is

TABLE 3 | A selection of frequent, multi-action episodes in the East African cheetah, and South African rhino actions history data sets.

Episode	Frequency	Position in sequence	Action
		East African cheetah	
1	15	1	Elephants_trample_crops
		2	Hippo_attacks_some_cattle
3	11	1	Donate_equipment_for_anti-poaching_enforcement
		2	Evict_residents_from_reserve
2	7	1	Seize_elephant_ivory
		2	Donate_equipment_for_anti-poaching_enforcement
		3	Elephants_trample_crops
		South African rhinos	
1	151	1	Sell_a_few_rhino_horns
		2	Sell_a_few_rhino_horns
2	9	1	Report:_large_animal_populations_are_decreasing
		2	Sell_a_few_rhino_horns
3	9	1	Sell_a_few_rhino_horns
		2	Sell_a_few_rhino_horns
		3	Shoot_poaching_suspects

Episodes are found with the PPS algorithm of Ma et al. (2004).

on the development of a new relational database of political-ecological actions that have been taken from original, non-database sources.

4.3. Constructing Ecosystem Management Tools

In addition to its role in developing political-ecological theory, an EMAT database is engineered to support existing and future ecosystem management decision support tools. How EMAT databases are, and should be employed in these two support roles is described in the following sections.

4.3.1. Supporting Existing Tools

Efforts are underway to build political-ecological system models that represent the effect of different ecosystem management policies. If such models were statistically fitted to political-ecological data, their use in assessing proposed ecosystem management options would be more credible (Haas, 2020). For example, Haas and Ferreira (2018) develop an agent-based model of the political-ecological system that contains the South African rhino population. As these authors state: "One part of this model is a submodel of poachers interacting with the rhino population. Combinations of antipoaching initiatives and economic opportunities are evaluated as to their probable effectiveness at changing local people's inclination to poach rhinos and the consequent effect on the rhino population." These authors first fit their model to a political-ecological data set composed of coupled observations on poaching actions, antipoaching actions, and a `collect data` set of rhino abundance observations. Then, a particular management policy is evaluated by running the fitted model under this policy and examining its effect on rhino abundance.

As another example of an existing tool, Miyasaka et al. (2017) develop an agent-based, social-ecological model of land use by farmers in Inner Mongolia. They use this model to study the impact of different land management policies on the impacted dryland ecosystem. Their model is spatially-explicit, incorporates a learning mechanism, and enacts two-way interaction with the ecosystem (anthropogenic effects on the ecosystem and feedback effects by the ecosystem on the farmers). Their model also allows for time lags. They empirically calibrate the farmer submodel with survey data, and the ecosystem submodel with biophysical measurements. If their social-ecological data were entered into an EMAT RDBMS, model calibration exercises could be repeated in a transparent way as new data became available. Or, more impactfully, such an RDBMS could support the statistical fitting of the model's parameters with (say) maximum likelihood. This RDBMS would make the many database queries needed to accomplish this model-wide statistical estimation exercise easier to organize, perform, and communicate.

4.3.2. Supporting Future Tools

Kupschus et al. (2016) call for *integrated monitoring programmes* to support the ecosystem approach to conservation management. These authors argue that a monitoring program needs to provide the means to test causal relationships between anthropogenic actions and ecosystem responses. Such monitoring would produce large amounts of political-ecological data. This data would need to be housed in an accessible way. In a similar vein, Ascough et al. (2008) identify challenges to assessing models used to support environmental and ecological decision-making. These authors see a need for a "holistic, integrated uncertainty framework" in order to comprehensively incorporate uncertainty into environmental decision making. They propose a web-based system as one way to implement such a framework.

```
report prepare_data
   rdbms(east-af-cheetah efgroups.dat efregions.dat efprsd.dat
   declare global temporary table v (storyid int, senid int,
      mwvrb varchar(30000), action varchar(80)) not logged
   endcommand
   insert into session.v
   SELECT distinct subvs.storyid, subvs.senid, subv.mwvrb, action
   FROM (SELECT mwvrb.vrbuid, mwvrb.mwvrb, emat.action
         FROM mwvrb, ematvrbemat
         JOIN emat ON emat.ematid = ematvrbemat.ematid
         JOIN ematvrb ON ematvrb.vrbuid = ematvrbemat.vrbuid
         WHERE mwvrb.mwvrb = ematvrb.ematvrb
         AND (emat.action IN ('open_reserve_to_settlement',
               'reduce_game_reserve_size',
               'sell_grazing_land_to_individuals',
               'destroy_lake_from_farming_and_industry',
               'block_migration_routes_with_settlements',
               'build_road_through_NP', 'clear_new_land'))
        ) subv
   JOIN (SELECT sen.storyid, mwvrbsen.senid, mwvrb
         FROM mwvrbsen
         JOIN mwvrb ON mwvrb.senvrbid = mwvrbsen.senvrbid
         JOIN sen ON sen.senid = mwvrbsen.senid) subvs
   ON subv.mwvrb = subvs.mwvrb
   endcommand
   declare global temporary table d (storyid int, senid int,
      doph varchar(30000), action varchar(80)) not logged
   endcommand
   insert into session.d
   SELECT distinct subds.storyid, subds.senid, subd.doph, action
   FROM (SELECT doph.dophuid, doph.doph, emat.action
         FROM doph, ematdophemat
         JOIN emat ON emat.ematid = ematdophemat.ematid
         JOIN ematdoph ON ematdoph.dophuid = ematdophemat.dophuid
         WHERE doph.doph = ematdoph.ematdoph
         AND (emat.action IN ('open_reserve_to_settlement',
               'reduce_game_reserve_size',
               'sell_grazing_land_to_individuals',
               'destroy_lake_from_farming_and_industry',
               'block_migration_routes_with_settlements',
               'build_road_through_NP', 'clear_new_land'))
        ) subd
   JOIN (SELECT sen.storyid, dophsen.senid, doph
         FROM dophsen
         JOIN doph ON doph.sendophid = dophsen.sendophid
         JOIN sen ON sen.senid = dophsen.senid) subds
   ON subd.doph = subds.doph
   endcommand
   SELECT story.storydate, v.action
   from session.v v
   JOIN session.d d
   ON v.storyid = d.storyid AND v.senid = d.senid AND v.action = d.action
   JOIN story ON story.storyid = v.storyid
   endquery
   build)
```

FIGURE 6 | An `rdbms_()` relation to build temporary tables in the `east-af-cheetah` EMAT database and then query them for habitat-destroying EMAT actions.

These authors envisage a system that contains two databases: One holding ecosystem observations, e.g., geospatial data, and one holding output from models of ecosystem function. These databases would be used to compare the quality and precision of different ecosystem models. Quality would consist of side-by-side comparisons of reliability and validity; and precision would consist of side-by-side comparisons of values on uncertainty measures, e.g., prediction intervals of a biodiversity index across time. These comparisons would be used by both scientists and policymakers to select models to use to inform what actions to take to manage an at-risk ecosystem. The two constituent databases would contain large numbers of interconnected variables and many multivariate observations on them. Although the authors are silent on the nature of these databases, their vision for how data would be used by the system points to a need for databases of ecosystem observations that are as comprehensive and accessible as possible. A single EMAT RDBMS would be an accessible portal capable of holding enough variables and observations to achieve such comprehensiveness. This would be done by populating the database with pairs of observations on the `collect data` EMAT action—one member of a pair having the `source` attribute set to *observed*, and the other with `source` set to *model*.

4.4. Conclusions

This article has demonstrated that data on taxonomy-indexed political-ecological actions can be organized into an EMAT RDBMS implemented within a single, stand-alone Java program.

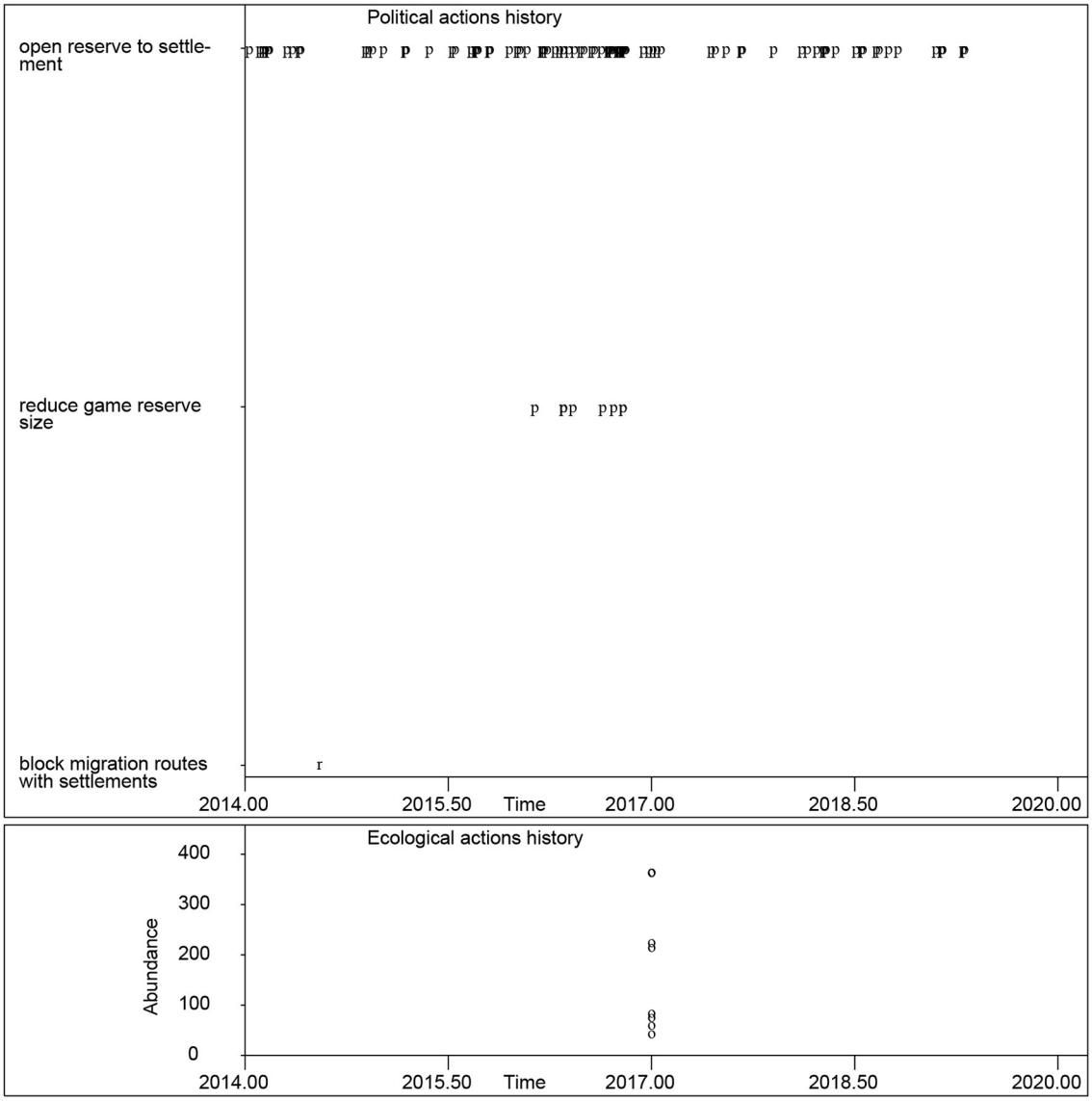


FIGURE 7 | Actions history of those actions that contribute to the destruction of cheetah habitat. See **Figure 4** for the legend.

TABLE 4 | Episodes in the East African cheetah database that contain the habitat-destroying EMAT action of `open reserve to settlement`.

Episode	Frequency	Position in sequence	Action
1	6	1	Invest_in_tourism_infrastructure
		2	Open_reserve_to_settlement
2	8	1	Open_reserve_to_settlement
		2	Seize_elephant_ivory

The exercise of building a relational database for political-ecological data illustrates how the requirement that database tables be linked forces the researcher to hypothesize about how different entities and components of political-ecological actions might be related to each other. At this early stage in our understanding of how groups and ecosystems react to different policies, such an exercise can only be beneficial.

Given the continuing decline in global biodiversity (Ceballos et al., 2015), there is an immediate need for a larger group of people to become involved in ecosystem management. Social scientists need to work with ecologists, business leaders, and government officials to quickly find effective policies to manage the interface between humans and ecosystems. The database technology described herein, if implemented on data taken on the planet's most at-risk ecosystems, would help to bridge these gaps. In order to do this, however, these databases will need to be connected to persistent streams of political-ecological data.

4.5. Next Steps

4.5.1. Shortcomings

EMAT action equivalence sets are manually developed and have been completed for only a small subset of the actions in the EMAT. This step in the parsing preprocessor needs to be automated with current parsing algorithms that reference a large corpora of international stories. Complex queries that are run against large EMAT databases can be computationally expensive. These databases and queries need to be in a RDBMS that is running on a high performance computing system. The porting and tuning required to achieve acceptable runtimes on such systems is a nontrivial task and further, not all researchers will have access to such systems. Maintaining an EMAT database that is connected to online story feeds requires an investment in hardware and trained staff. This resource requirement may keep many researchers from developing their own EMAT databases.

4.5.2. The Future

The work of the present article suggests two research directions. The first is to compile a list of all current political-ecological models and then update the EMAT database schema so that its entities, attributes, and linkages represent the observable variables of as many of these models as possible. Doing so would allow a common language of political-ecological theory to evolve as called for by Frey and Cox (2015). Second, an EMAT database needs to be built for each ecosystem that hosts species who are at risk of extinction. These databases would then be used to find management policies that conserve them.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

TH developed the theory, wrote all computer code, created all figures and tables, and wrote the article.

SUPPLEMENTARY MATERIAL

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A "Wicked Problem" Reconciling Human-Shark Conflict, Shark Bite Mitigation, and Threatened Species

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Conservation measures often result in a "wicked problem," i.e., a complex problem with conflicting aims and no clear or straightforward resolution without severe adverse effects on one or more parties. Here we discuss a novel approach to an ongoing problem, in which actions to reduce risk to humans, involve lethal control of otherwise protected species. To protect water users, nets are often used to catch potentially dangerous sharks at popular bathing beaches, yet in Australian waters one of the targeted species, the white shark (*Carcharodon carcharias*) is listed as Vulnerable, while bycatch includes the Critically Endangered grey nurse shark (*Carcharias taurus*). Recent, highly publicised, shark attacks have triggered demands for improved bather protection, whilst welfare and conservation organisations have called for removal of lethal measures. This leaves management and policy makers with a wicked problem: removing nets to reduce impacts on threatened species may increase risk to humans; or leaving the program as it is on the premise that the benefits provided by bather protection are greater than the impact on threatened and protected species. We used multivariate analysis and generalised additive models to investigate the biological, spatial-temporal, and environmental patterns influencing catch rates of threatened and of potentially dangerous shark species in the New South Wales shark nets over two decades to April 2019. Factors influencing catches were used to develop a matrix of potential changes to reduce catch of threatened species. Our proposed solutions include replacing existing nets with alternative mitigation strategies at key beaches where catch rate of threatened species is high. This approach provides stakeholders with a hierarchy of scenarios that address both social demands and threatened species conservation and is broadly applicable to human-wildlife conflict scenarios elsewhere.

Keywords: human-wildlife conflict, shark, wicked problem, conservation, threatened species

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INTRODUCTION

A human-wildlife conflict is characterised by animals posing a direct and recurrent hazard to the livelihood or safety of people, which in turn leads to the persecution of the species involved (IUCN SSC, 2021). These scenarios are frequently composed of a complex interplay of ecological, social, and climate factors, involved in forcing wildlife and people to co-occur and share limited resources (Abrahms, 2021). Human-wildlife conflict is a recurrent topic in conservation biology as

management actions have implications for both the people and the animals involved (Dickman, 2010). While most research has generally approached the ecological and human dimensions of human-wildlife conflicts separately, identifying the linkages between both processes has been identified as a powerful strategy toward mitigation of these clashes (Teixeira et al., 2021). Decisions tend to be anthropocentric (i.e., favouring humans over animals), and mostly focused at current interests rather than considering its future significance for the people involved (Treves and Santiago-Ávila, 2020). While humans frequently underestimate or neglect risk from wild animals, we also show very little to no tolerance for it (Treves and Bruskotter, 2014). A risk-based approach to tackle human-wildlife conflicts means that decisions should be made based on the likelihood and consequence of these events, instead of focused on its probability of occurrence (Hudenko, 2012; Lischka et al., 2020).

Shark attacks on humans are the most global of all human-wildlife conflicts since they occur in all ocean basins and around all continents of the planet, bar Antarctica (Hardiman et al., 2019). They have received increasing attention worldwide following an apparent rise in the rate of incidents over the last three decades (Midway et al., 2019; Ryan et al., 2019). While shark attacks remain rare (∼100 per year globally), they evoke powerful responses from both the public and government agencies as they often result in severe injuries or death (Sabatier and Huveneers, 2018). Approaches to mitigating shark-human interactions have historically relied on killing sharks in the area. Since 1937 this approach has used nets deployed off beaches to reduce local and/or transient populations of potentially dangerous sharks and thereby minimise the likelihood of interactions with people (Reid and Krogh, 1992; Dudley and Cliff, 2010; McPhee et al., 2021). However, a variety of non-targeted species are also bycaught in these programs (Krogh and Reid, 1996; Cliff and Dudley, 2011), raising community concerns about possible unintended broader ecosystem impacts.

Inevitably, management conundrums arise from addressing human-wildlife conflicts, which usually encompass ecological, economic, and socio-political groups with differing levels of decision-making power and values. As a result, proposed measures to address the conflict lead to resolutions satisfactory for some stakeholders yet to the detriment of others (Redpath et al., 2013; Bunnefeld et al., 2017). These are known as wicked problems because the complexity of these conflicts makes reaching ecologically desirable conservation outcomes challenging and precludes simple and well-defined solutions (Rittel and Webber, 1973). In a wicked problem, any proposed solution for the issue will generate a new, often different, problem, thus influencing any resolution of the solution and creating new difficulties (Game et al., 2014). Management needs to mitigate shark attacks are in many ways more driven by community fear of an incident than the actual risk, and provides management with a classic wicked problem, as nets deployed to protect bathers pose a threat to non-dangerous shark species and other marine taxa and their removal evokes strong, divergent public opinions.

The Shark Meshing Program (SMP) in New South Wales (NSW), Australia, is the longest-running initiative to reduce shark-human interactions in the world (Reid and Krogh, 1992). Claimed to be an affordable and effective practise to minimise shark peril (NSW Shark Menace Committee, 1929), it has run since 1937 and specifically targets three potentially dangerous sharks, white (*Carcharodon carcharias*), tiger (*Galeocerdo cuvier*), and bull (*Carcharhinus leucas*) sharks (Lee et al., 2018). It currently comprises 51 beaches and in a typical year, e.g., 2016–2017, captured 373 animals of which only 7.5% comprised target species (22 white, 3 tiger and 3 bull sharks) (Lee et al., 2018). The capture of species that are listed as threatened under NSW legislation, i.e., Fisheries Management Act 1994 (FM Act) and Biodiversity Conservation Act 2016 (BC Act); leads to a complex situation, with the program listed as a Key Threatening Process under both the FM Act and the BC Act because it threatens the survival of numerous threatened species, including white and grey nurse sharks (*Carcharias taurus*). Grey nurse sharks use predictable and well-documented aggregation sites, many of which have been given regulatory protection with site-specific fishing rules (Lynch et al., 2013). Even though the species has been protected since 1984 (Pollard et al., 1996), mortality arising from interactions with fishing gear appears to still be inhibiting their recovery (Robbins et al., 2013).

Mitigation of this Key Threatening Process could most readily be achieved by complete removal of nets, however, this may have perverse consequences. Beach goers and lifesavers are the people in the water exposed to potentially dangerous sharks, so they will be directly affected by any changes made to the program by the NSW state government and are therefore key stakeholders in any decision-making process. Beach authorities such as life savers are inevitably the first responders at shark attacks and, in fact, were the primary motivators for shark nets to be deployed in NSW following spates of shark-related severe injuries and fatalities (Conrick, 1989). Similarly, local councils who administer the facilities and beaches where nets are placed, receive substantial income from visitors using these amenities and are therefore important stakeholders. Although shark nets have reduced shark attacks by 88–91% in regions that they have been deployed (Dudley, 1997), there is substantial bycatch of harmless marine animals (Krogh and Reid, 1996; Cliff and Dudley, 2011; Dalton and Peddemors, 2018). Environmentally conscious public and organisations are therefore also important stakeholders in determining the future of shark nets. However, if nets were removed wholesale, even a single human fatality at a previously meshed beach, could trigger a negative public and political reaction, which may lead to calls to immediately reinstate the program and/or for renewed culling (Pawle, 2017) with possible adverse consequences for threatened species. This leads to the SMP falling into the category of a wicked problem, as the primary target species of the program, white sharks, are themselves a threatened species. While government and legislators are tasked with protecting this species to ensure their survival, they are simultaneously required to protect people from this protected species. Protection of human life is the highest priority for government, mandated by multiple regulations and legislation, yet retention of the status quo is inconsistent with conservation legislation that aims to abate, ameliorate, or eliminate the adverse effects of Key Threatening Processes.

Resolving wicked problems requires ingenuity combined with clear evidence of drivers of conflict (Guerrero et al., 2017; Mason et al., 2018), and an educated and supportive public and political community. Currently there are several alternatives to shark nets (McPhee et al., 2021), however, these might be more expensive and/or labour intensive and are not necessarily supported by the general public (Simmons and Mehmet, 2018). Replacing nets by more selective fishing gear such as a series of anchored buoys with baited hooks, known as "drumlines," on which the hooked shark fights to fatal exhaustion, has been previously found to be an effective strategy to reduce the impacts of mitigation programs while still promoting beach safety (Cliff and Dudley, 2011). However, none of the available mitigation strategies can completely stop sharks from using a particular area (Guyomard et al., 2020) and the presence of potentially dangerous sharks might not necessarily equate to risk of shark attack. Here we use 22 years of empirical data on catch rates of sharks in the SMP to identify trends in capture of non-targeted threatened (i.e., grey nurse sharks) and targeted (i.e., white, tiger, and bull sharks) species and localised drivers of shark presence. We use the model outcomes to predict how possible changes in the operation of the SMP may affect both shark conservation and human safety. Our main objective is to reduce captures of grey nurse and white sharks for conservation purposes while maintaining low risk of shark interactions to humans. However, this is complicated by white sharks being one of the potentially dangerous species, wherein lower captures of this species could potentially lead to a higher risk of shark bites. By proposing a hierarchy of alternatives to shark nets, focused on the locations with higher captures of threatened species, we provide stakeholders with information by which to choose relative changes for threatened species conservation vs. perceived risk of shark bite. The methods adopted in this study may be applicable for addressing other human-wildlife conflict scenarios elsewhere.

MATERIALS AND METHODS

Shark Meshing Program and Catch Data

The study area comprises ∼190 km of the 2,137 km NSW coastline (**Figure 1**) that encompass three major coastal cities and a total of 51 beaches with a single shark net set off each beach. Shark nets are 150 m long and 6 m in height and set in the bottom half of the water column on the sandy seabed approximately 500 m offshore in 10–12 m depth water (Krogh and Reid, 1996). Catch data used for analyses was provided directly by the NSW government to the research team and spanned 22 years from 1998 to 2019. Throughout this time, nets were only in the water for 8 months of the year (01 September to 30 April) as they are removed over winter to reduce potential whale entanglements during historically low months of beach use by swimmers, albeit wetsuits enable surfers to surf year-round. During the first 11 years of sampling, i.e., 1998–2008 (hereafter Period 1), nets were deployed for a minimum of 9 weekdays and every weekend per month. During this period, the maximum time that nets could fish without being checked was 4 days (96 h). From January 2009 onwards (hereafter referred as Period 2) nets were continuously in the water, also from September to April, but checked every 72 h as an effort to reduce mortality of captured individuals (Reid et al., 2011). Detailed descriptions of the SMP methodology can be found in Reid et al. (2011). Live sharks were sexed, measured to fork length and released. Maturity of sharks was determined using published sex-specific maturation sizes (Kohler et al., 1996; Lucifora et al., 2002; Cruz-Martínez et al., 2005; Holmes et al., 2015).

Among the 51 beaches meshed within the SMP several are in close proximity. To avoid bias in catch data, we grouped nets that were within 1 km of each other into a single netted area, producing 41 independent netted areas for our analyses (**Figure 1**). This grouping resulted in uneven effort, e.g., areas with two or more nets vs. single net areas (**Supplementary Table 1**). Therefore, fishing effort was standardised to metres of net per fishing day (i.e., 4-day for Period 1 and 3-day for Period 2).

Total catch data comprised only the days in which a net was checked and at least one individual from any of the four shark species captured, so we conducted a second standardisation for analysing fine-scale species-specific catch rates similar to Lee et al. (2018). As nets were checked on Monday/Tuesday and Friday prior to 2009, fine-scale effort (i.e., actual days with fishing effort) could not be accurately determined for Period 1. From 2009, cheques occurred every Monday, Wednesday, and Friday, enabling subdivision of Period 2 into three weekly time intervals (Monday and Tuesday; Wednesday and Thursday; Friday to Sunday) of fishing effort. Catch data were matched using reported dates of capture. All catches of the four shark species are included in the analysis as mortality rates of target sharks in the nets are 95% for Period 1 and 86% for Period 2 and, although live sharks are tagged and released, post-release survivorship is unknown (Dalton and Peddemors, 2018). The impacts of the SMP were only assessed for the two threatened shark species captured, i.e., grey nurse and white sharks.

Physical and Environmental Variables

Seasonality in SMP catches was related to species-specific migratory patterns along the east coast of Australia using the 8 months of fishing effort. The influence of physical characteristics of netted areas influencing shark captures in the nets were assessed using: (i) proportion of each beach that is covered by a net (net:beach) (%); (ii) percentage of rocky reef cover within a 1 km^2 quadrat centred on net location, downloaded from the NSW Government database (NSW Government, 2019); and (iii) distance to nearest known grey nurse shark aggregation site (**Figure 1**) to assess whether the species would be more vulnerable to capture in the nets deployed near to where these aggregations occur (Barker and Williamson, 2010). The environmental influence of sea surface temperature (SST; °C) was also tested (Wintner and Kerwath, 2018). Daily values of SST with 0.02° spatial resolution were downloaded from the Australian Ocean Data Network Portal (AODN, 2019) and used to calculate mean values per set during Period 2. The nearest quadrats of data available from the corresponding netted areas were used. To standardise habitat of netted areas with multiple net-sets, a 1 km^2 quadrat was created around the nets for each area. The percentage of reef area covered in this polygon and distance from

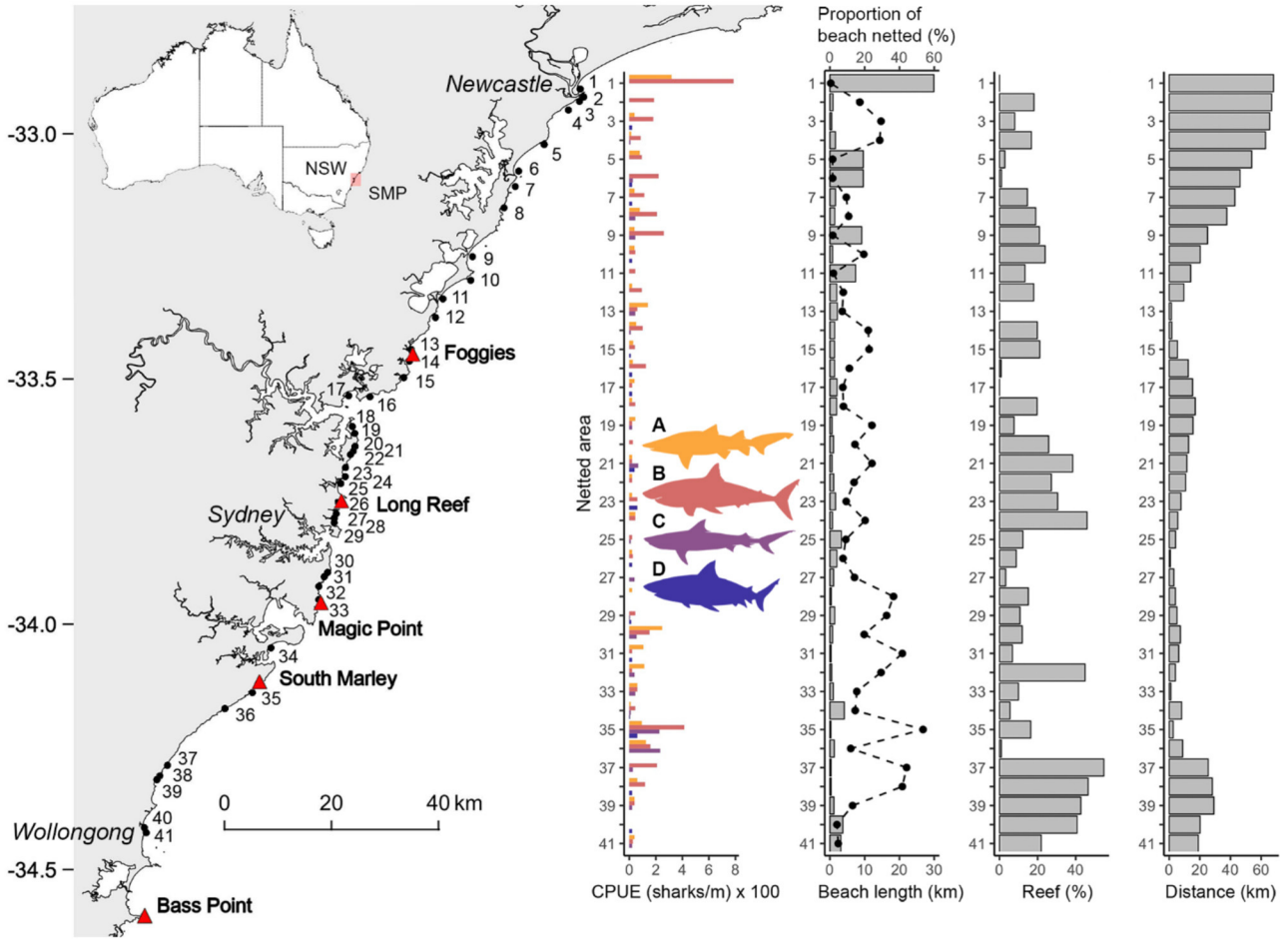


FIGURE 1 | Map of the area covered by the Shark Meshing Program (SMP) showing the locations of the known aggregation sites of grey nurse sharks (triangles) and the 41 netted areas (circles), including total catch-per-unit-of-effort (CPUE) of **(A)** grey nurse, **(B)** white, **(C)** tiger, and **(D)** bull sharks between September 1998 and April 2019. The distributions of the physical parameters beach length, proportion of beach netted, percentage of reef area (Reef), and their corresponding distance from the nearest grey nurse shark aggregation site (Distance) are also represented.

its midpoint to the nearest grey nurse shark aggregation site, were then included in the analysis. All physical variables were calculated using QGIS software (version 3.4.0).

Shark Occurrence Patterns

Data spanning the entire study period was investigated using a partial redundancy analysis (RDA). Each species was divided into sex-specific groups and maturation status prior to analysis. To account for the differences in net checking regimes between Period 1 and Period 2, a new categorical variable named SMP-period divided shark catches into before and after 2009. The RDA full model also included Area as a geographic component, given SMP distribution is effectively latitudinal (**Figure 1**), and the corresponding physical parameters of netted areas.

Zero-inflated generalised additive models (ZIGAM) were used to model shark fine-scale occurrence patterns. Models were calculated individually for each species, and the logarithm of fishing effort included as an offset covariate. Groups of continuous candidate predictor variables included: (i) spatial-temporal: netted area and month and (ii) environmental: SST, proportion of beach netted, proportion of reef area cover, and distance to nearest grey nurse shark aggregation site. Interactions between variables were tested to investigate whether complex relationships would better explain shark catch in the SMP. Multicollinearity was assessed with Pearson's correlations. Total number of sharks caught per net deployment were included as response variables using zero-inflated Poisson families of error distribution. Starting from the null models, new variables were gradually included to identify lower AIC values (Wagenmakers and Farrell, 2004). Final models were inspected for normal residual distribution.

Addressing the Wicked Problem

To determine whether alternatives to shark nets could assist in shark conservation whilst still affording protection from being bitten by a shark, differences in captures of each species were investigated considering potential scenarios of removing particular netted areas from the SMP. For this purpose, the

significant variables found to influence catches of the threatened species (i.e., grey nurse and white sharks) or modifying the netted areas with the highest captures of grey nurse sharks, were used. The values of each covariate responsible for a higher likelihood of capture were used to estimate reductions in catch rates for each species. Each scenario is hereafter referred to as potential changes, and the amount of catch reduction per species compared using ANOVA with *post-hoc* Tukey tests.

The effects of each potential change upon vulnerability to capture for each shark species were compared based on productivity and susceptibility scores, which are continuous indices ranging from zero to three. This approach is similar to what is used in fisheries research to assess stock vulnerability to becoming overfished (Stobutzki et al., 2001; Patrick et al., 2009; Gallagher et al., 2014). The productivity score has a biological meaning and relates to the capacity of a shark species to produce maximum sustainable yields and to recover if its population is being depleted (Patrick et al., 2009), whereas here the susceptibility score relates to the potential of a species to be impacted by their captures in the SMP. Species-specific productivity scores were obtained from Patrick et al. (2009). Susceptibility scores were calculated based on the average of the productivity score and the following attributes: (i) seasonal migrations: number of months present within the SMP region in relation to total meshing period, i.e., months captured against months with SMP effort; (ii) spatial overlap: overall area used when present in relation to the total 41 meshed areas, i.e., overlap between northernmost and southernmost locations for each species from telemetry studies and the SMP net deployments; (iii) mortality: number of alive vs. dead sharks from each species in the SMP since 2009 (i.e., the last modification of the program); (iv) desirability: 0 = non-targeted and 3 = targeted; (v) management strategy: 0 = no recovery plan in place in Australian waters and 3 = recovery plan in place in Australian waters; (vi) SMP impact: 0 = SMP is not a Key Threatening Process and 3 = SMP is a Key Threatening Process; and (vii) population impact: maximum number of individuals caught per year in the SMP in relation to the East Australian population of the species (**Supplementary Table 2**). The attributes were first calculated as percentages from either the SMP or published data (Lincoln-Smith and Roberts, 2010; Otway and Ellis, 2011; Tillett et al., 2012; Holmes et al., 2017; Hillary et al., 2018; Bruce et al., 2019; Lee et al., 2019), and then standardised to the same 0–3 scale used in Patrick et al. (2009) for comparison purposes. The catch reduction rates for each species from potential changes to the SMP were similarly converted to this scale in order to be also included as attributes for the final vulnerability scores, calculated using the formula:

$$Vulnerability\ score = \bar{x}(X,\ Y)\ x\ Z$$

In which X is the susceptibility score for the species in the SMP, Y is the scaled catch reduction rate, and Z is the percentage fishing effort reduction (i.e., number of nets removed from the current total 51 nets in the SMP) for the corresponding potential change (**Supplementary Table 2**). The significance level of the vulnerability score variations as a function of the potential changes were assessed using ANOVA.

A wicked problem matrix was developed for our case study of the SMP, providing a series of hypothetical potential changes that could be implemented to the program. These ranged from leaving the status quo, through to selective removal of nets potentially combined with alternative mitigation options, to completely removing all shark nets without replacement with any other mitigative measures. Expected consequences of potential actions for some key stakeholder groups involved in the SMP context (Fraser-Baxter and Medvecky, 2018; Simmons and Mehmet, 2018; SharkSmart NSW, 2021) and for the shark species were graded from low to high resistance and risk.

Five main stakeholder groups were identified: (i) Non-Government Organisations (NGO): supporting conservation through non-lethal measures, (ii) beach goers: people directly affected by any changes to the SMP, (iii) lifesaving associations (volunteer and professional): tasked with protecting beach goers and inevitably, the first responders in the case of a shark attack, (iv) local councils: potential beneficiaries of shark bite mitigation measures, and (v) NSW government: through the Department of Primary Industries Fisheries, is the government agency that administers the SMP. Five groups of primary actions were identified: (i) status quo: leaving the program as is, (ii) selective removal: removing nets as identified in **Table 1**, (iii) selective replacement: replacing some nets with alternatives, (iv) total replacement: fully removing nets and replacing with alternatives, and (v) completely removing any shark bite mitigation measures (**Table 2**). Alternatives were identified through stakeholder questionnaires as being the most strongly supported alternative measures as described in the current literature (Crossley et al., 2014; Gray and Gray, 2017; Simmons and Mehmet, 2018). These include Shark Management Alert in Real-time (SMART) drumlines (Guyomard et al., 2019; Tate et al., 2019), traditional drumlines (Sumpton et al., 2011; McPhee et al., 2021), helicopters (Crossley et al., 2014), drones (Butcher et al., 2019), listening stations (VR4Gs) for detecting tagged sharks and informing the public through smartphone apps in real-time (Simmons and Mehmet, 2018), and different combinations of these methods (**Table 2**). The alternatives are not an exhaustive list but examples of approaches that could be adopted based on current use in Australia and elsewhere (McPhee et al., 2021).

For the proposed changes, resistance levels for stakeholder groups were graded according to a traffic light scale based on results from open and targeted online questionnaires and forums, recently developed as part of the community engagement and social research by the NSW Department of Primary Industries, to better understand the opinions and attitudes of people toward coexisting with sharks (SharkSmart NSW, 2021; Simmons et al., 2021). Since the vulnerability scores calculated for each species are also returned using a traffic light scale, and directly included the corresponding scaled catch reduction rates of each proposed change, these were employed for the grading system of risk for sharks. Finally, the risk level of alternative mitigation strategies to each shark species were based on the literature as to whether they could lead to higher catch rates (i.e., high risk), lower catch

TABLE 1 | Potential changes to the Shark Meshing Program fishing effort, based on the significant variables identified by the fine-scale modelling approach.

Change	Areas to be removed	Number of nets removed	Reduction in effort (%)	Expected catch reduction				T
				Grey nurse	White	Tiger	Bull	
No removal (status quo)			0	0	0	0	0	A
Remove Stockton	1	1	1.96	15.43	18.44	0	0	B
Remove Bondi	30	1	1.96	11.89	3.59	5.43	0	B
Remove Stockton and Bondi	1, 30	2	3.92	27.32	22.04	5.43	0	B
Remove areas with >40% of reef area	24, 32, 37, 38, 39, 40	6	11.76	12.54	10.16	8.84	10.30	B
Remove areas 29–33		7	13.72	25.39	6.98	14.96	8.58	B
Remove areas 1–6		8	15.68	21.97	36.36	3.63	10.30	B
Remove areas with >30% of proportion of beach netted	28, 29, 31, 35, 37, 38	8	15.69	13.50	18.86	26.51	29.18	B
Remove areas 1–9		9	21.57	29.68	49.96	13.14	15.45	C
Remove areas 33–41		11	23.53	20.81	25.16	56.94	27.04	C
Remove areas with <10% of proportion of beach netted	1, 5, 6, 7, 9, 11, 12, 13, 17, 18, 23, 25, 26, 40, 41	16	31.37	37.61	43.92	15.29	41.20	C
Remove areas with <10% of reef area	1, 3, 5, 6, 13, 16, 17, 19, 26, 27, 31, 33, 34, 36	17	33.33	49.10	42.67	40.62	32.19	C
Remove all nets		51	100	100	100	100	100	D

Included are the respective percentage reductions in effort originated from each change, expected catch reductions for each threatened shark species and the significant grouping identified by the post-hoc Tukey test (T).

rates (i.e., low risk). When information was not available and could not be fully determined it was scored as a medium risk. The significance levels were set at 0.05 and all statistical analyses were performed in R (version 4.0.5).

RESULTS

Total Shark Catch

A total of 106 grey nurse, 217 white, 48 tiger, and 22 bull sharks were confirmed as caught during the study. Bull sharks were historically included in a "whaler shark" category and have only been identified to species level since 2010 (Lee et al., 2019), partly explaining the lower numbers caught in comparison to the other species, therefore bull shark species-specific data were only available for Period 2. More sharks were caught during Period 2 (72.5%). The total number of sharks found alive when nets were checked, and subsequently released, increased from 12.0% during Period 1–40.3% in Period 2. The total bull shark catch was low but equally distributed along the SMP region (**Figure 1**). Captures for the other species were widespread throughout the SMP region with localised peaks (**Figure 1**). Captures of grey nurse sharks were highest at Stockton Beach (Area-1; 0.7 sharks/year), followed by Bondi (Area-30; 0.5 sharks/year). Terrigal Beach (Area-13) was the only location near a known grey nurse shark aggregation site (0.7 km) with slightly higher captures of the species (**Figure 1**) but it still only had an average catch rate of 0.3 sharks/year. White shark catch was also highest at Stockton Beach (1.6 sharks/year), followed by Wattamolla (Area-35; 0.8 sharks/year) (**Figure 1**). Tiger sharks were mostly caught at Wattamolla (0.4 sharks/year) and Garie (Area-36; 0.5 sharks/year) beaches (**Figure 1**).

Catches of grey nurse sharks were dominated by females (female:male = 5.93; **Figure 2A**). More even sex ratios were observed for white (female:male = 1.14; **Figure 2B**), tiger (female:male = 1.09; **Figure 2C**), and bull (female:male = 1.71, **Figure 2D**) sharks. While most grey nurse, tiger, and bull sharks caught were mature, white sharks were almost entirely juveniles with only 5 (2.3%) mature sharks caught during the entire study period (**Figure 2**). According to Dudley and Cliff (2010), bull sharks larger than 175.5 cm fork length are capable of severe tissue removal and should be considered as potentially hazardous to humans. Most white, tiger, and bull sharks caught in the SMP were indeed larger than 175.5 cm fork length (**Figure 2**).

Broad-Scale Occurrence Patterns

The final RDA model revealed that the variables Maturation Status ($F = 272.8$; $p = 0.005$), month ($F = 13.9$; $p = 0.005$), SMP-period ($F = 9.7$; $p = 0.005$) and Area ($F = 2.5$; $p = 0.041$) significantly influenced the total shark catches in the SMP (variance explained = 46.8%). While most groups showed similar trends, the patterns of female grey nurse, and male white and tiger sharks varied significantly (**Figure 3**). Female grey nurse sharks were predominantly adult and occurred more frequently during Period 2 (**Figure 3**). Similarly, male juvenile white sharks were more frequent during Period 2, but also during the first months of netting and in the northern area of the SMP (**Figure 3**). By contrast, male tiger sharks were caught toward the end of the netting season, especially during Period 1 (**Figure 3**).

Fine-Scale Occurrence Patterns

The capture of grey nurse sharks in the nets was influenced by interactions between netted area and month (deviance explained = 48.0%) and SST and proportion of reef substrate (84.6%)

TABLE 2 | Wicked problem matrix originated from the Shark Meshing Program, comprising a series of possible actions to the program and the expected outcomes for each stakeholder group and consequent risk for the shark species caught.

Stakeholders / Sharks	Low Resistance Risk	Medium Resistance Risk	High Resistance Risk

		Action		Stakeholders					Sharks			
Mitigation strategy	**Daily hours for gear deployment**	**Primary**	**Secondary**	**NGOs**	**Beach goers**	**Lifesavers**	**Councils**	**NSW Government**	**Grey nurse**	**White**	**Tiger**	**Bull**
Shark nets	24	Status quo							2.00	2.20	1.29	1.26
		Selective removal	1. Stockton						1.81	1.96	1.27	1.23
			2. Bondi						1.84	2.12	1.24	1.23
			3. Stockton + Bondi						1.66	1.88	1.21	1.21
			4. >40% reef						1.65	1.84	1.09	1.05
			5. Areas 29–33						1.51	1.83	1.03	1.04
			6. Areas 1–6						1.50	1.52	1.07	1.01
			7. >30% proportion of beach netted						1.57	1.68	0.95	0.91
			8. Areas 1–9						1.33	1.29	0.95	0.91
			9. Areas 33–41						1.37	1.47	0.71	0.83
			10. <10% proportion of beach netted						1.11	1.18	0.82	0.69
			11. <10% reef area						1.00	1.15	0.69	0.70
Drumlines	10	Selective replacement	SMART drumlines									
	24		Traditional drumlines									
Aerial surveillance	10		Drone contractors									
	1		Helicopter									
VR4G	24		Replacing some of the nets									
Aerial surveillance + drumlines	10		Drone + SMART drumlines									
	10		Helicopter + SMART drumlines									
Aerial surveillance + drumlines + VR4G	24		Drone + SMART drumlines + VR4G									
	24		Helicopter + SMART drumlines + VR4G									
Drumlines	10	Total replacement	SMART drumlines									
	24		Traditional drumlines									

(Continued)

TABLE 2 | Continued

Mitigation strategy	Daily hours for gear deployment	Primary Action	Secondary Action	Stakeholders: Sharks	NGOs	Beach goers	Lifesavers	Councils	NSW Government	Grey nurse	White	Tiger	Bull
					Low Resistance Risk		Medium Resistance Risk		High Resistance Risk				
Aerial surveillance	10		Drone contractors										
	1		Helicopter										
Aerial surveillance + drumlines	10		Drone + SMART drumlines										
	10		Helicopter + SMART drumlines										
Aerial surveillance + drumlines + VR4G	24		Drone + SMART drumlines + VR4G										
	24		Helicopter + SMART drumlines + VR4G										
VR4G	24	Removing all shark bite mitigating measures											

*Numbered cells for shark species represent the respective vulnerability scores calculated for each potential change from **Table 1**.*

(**Supplementary Tables 3**, **4**). Captures of grey nurse sharks had a northerly peak between September and January and a southerly peak in September (**Figure 4A**). The southern peak occurred between the aggregation sites of Long Reef and Magic Point, yet over 20 years only seven grey nurse sharks were caught at Area-13 (Terrigal Beach, 1.4 km from Foggies aggregation site) and three at Area-33 (Maroubra Beach, 1.3 km from Magic Point). Significantly lower grey nurse shark captures occurred between the aggregation sites of Foggies and Long Reef, particularly from January to April (**Figure 4A**). Overall, distance to nearest aggregation site did not significantly influence captures of grey nurse sharks in the SMP (**Supplementary Tables 3, 4**). However, captures of the species were related to cooler SSTs (<20°C) and exhibited a bi-modal topographic pattern, either more than 40% reef cover or when the seabed comprised <10% reef cover (i.e., sandy substrate) (**Figure 4B**).

White shark occurrence was influenced by the interaction between netted area and month (46.1%) and the environmental variables SST and proportion of beach netted (84.9%) (**Supplementary Tables 3, 5**). White shark catches exhibited a northerly peak between September and December and a southerly peak between January and April (**Figure 4C**). White sharks were more frequent when temperatures were cooler than 20°C (**Figure 4D**) particularly in areas with proportion of beach netted <10% and >30% (**Figure 4E**). Tiger shark catches were influenced by netted area (35.7%) (**Supplementary Tables 3, 6**), being mostly caught south of netted area 20 (**Figure 4F**).

Potential Changes to the SMP

We identified 12 scenarios where modifying the SMP could reduce captures of threatened sharks. These were divided into four significant groups according to the Tukey testing (**Table 1**). Removing all nets would reduce catches of grey nurse by 5.3 (±6.2) and white sharks by 9.8 (±7.3) sharks/year. It would, however, also reduce catches of tiger sharks by 2.5 (±1.3) and bull sharks by 2.4 (±1.8) sharks/year. On the other hand, removing nets from just Stockton, Bondi or both beaches (effort reductions between 2 and 4%; Tukey B) would lead to similar reductions to that achieved by reducing effort by 12–16% through removal of up to eight nets (**Table 1**). Stockton (3.5 ± 2.2 sharks/year) and Bondi (1.7 ± 0.8 sharks/year) beaches accounted for ∼30% of all grey nurse shark catches. Removing nets solely from Stockton would result in higher reductions in grey nurse shark catch and also lower the capture of white sharks, while removing Bondi alone would have a much less pronounced effect on white sharks (**Table 1**). Removing more nets (22 to 33% effort reduction; Tukey C) could reduce grey nurse shark captures by a maximum ∼50% (**Table 1**), but it would also greatly reduce captures of potentially dangerous sharks (**Table 1**), thereby feasibly increasing risk of shark bite if no alternative risk reduction measures are implemented.

The Wicked Problem Scenario

The potential changes would effectively reduce vulnerability scores for all species ($F = 37.91$; $p < 0.001$) (**Figure 5**). The status quo would maintain current impacts. At the other extreme, totally removing mitigation measures would completely

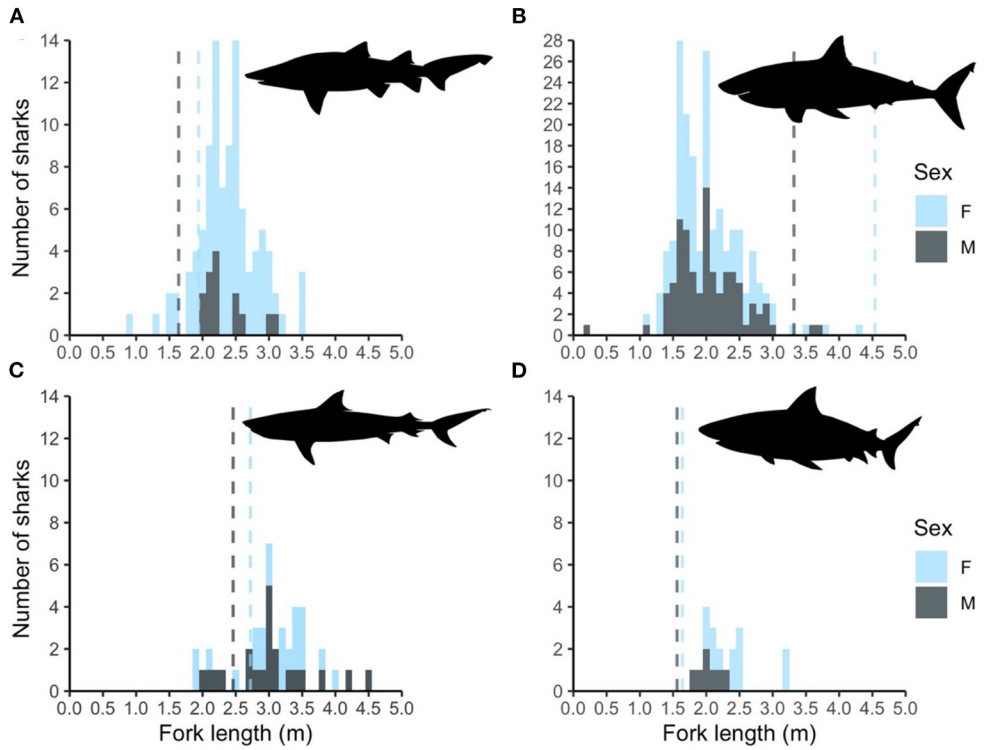


FIGURE 2 | Histogram of fork length distribution of the **(A)** grey nurse, **(B)** white, **(C)** tiger, and **(D)** bull sharks caught in the Shark Meshing Program between 1998 and 2019 by sex. Dashed lines represent sex-specific maturation sizes for each of the species obtained from the literature.

alleviate impacts from the SMP (**Table 2**; **Figure 5**). However, this might increase bather vulnerability, inviting a backlash if serious injuries or fatalities followed. Intermediate options include switching from nets to using more selective, or less lethal, fishing methods, but how this might influence catch rates is unknown as comparable data is not available.

Conventional drumlines that are operated in a manner that generally kills sharks are unpopular with the public (**Table 2**) (Gibbs and Warren, 2014). SMART drumlines significantly reduce mortality by providing a means of promptly responding to a catch (Guyomard et al., 2019). Selective replacement with SMART drumlines at specific beaches was the most acceptable modification to the SMP across all stakeholder groups (**Table 2**). However, many beaches are located further than the ∼15 km required to reach SMART drumlines and successfully release animals alive (Tate et al., 2019). Removal and replacement of the northern-most beach, Stockton, provides the most conservation outcome for the least reduction in effort, but simultaneously reduces the catch of what is arguably the primary target species of the SMP (i.e., white sharks), thus emphasising the wicked nature of this problem (**Table 1**; **Figure 4**). Stockton is adjacent to a port and outside of flight restriction zones, hence alternative mitigation measures could include SMART drumlines and drones (Simmons and Mehmet, 2018). Similarly, replacement of the Bondi net alone would provide substantial benefits to grey nurse sharks (**Table 1**).

DISCUSSION

We developed a hierarchical decision matrix of potential actions, based on catch trends, to reduce impact on threatened species whilst supporting bather safety in NSW (**Tables 1**, **2**). The conflict-to-coexistence continuum proposes that strategies to address human-wildlife conflicts should consider all possible actions from negative (i.e., with higher impacts for the animals) to positive (i.e., excluding risk for animals), and the intermediate more practicable solutions focused on coexistence (Frank, 2016). Our analysis allows stakeholders to apply science-based decision-making processes to formulate strategies for this human-wildlife conflict of a wicked problem incorporating threatened and target species.

Shark Occurrence Patterns

Climate change is increasing the frequency and severity of human-wildlife conflicts by forcing animals and people to share increasingly scarce resources (Abrahms, 2021). Most of the shark catches in the SMP occurred in more recent years (i.e., Period 2). While this could be associated with a higher fishing effort (i.e., nets spending more time in the water during Period 2 compared to Period 1), it may also be related to changes in distributions of potentially dangerous sharks along the East coast of Australia in response to climate change (Niella et al., 2020).

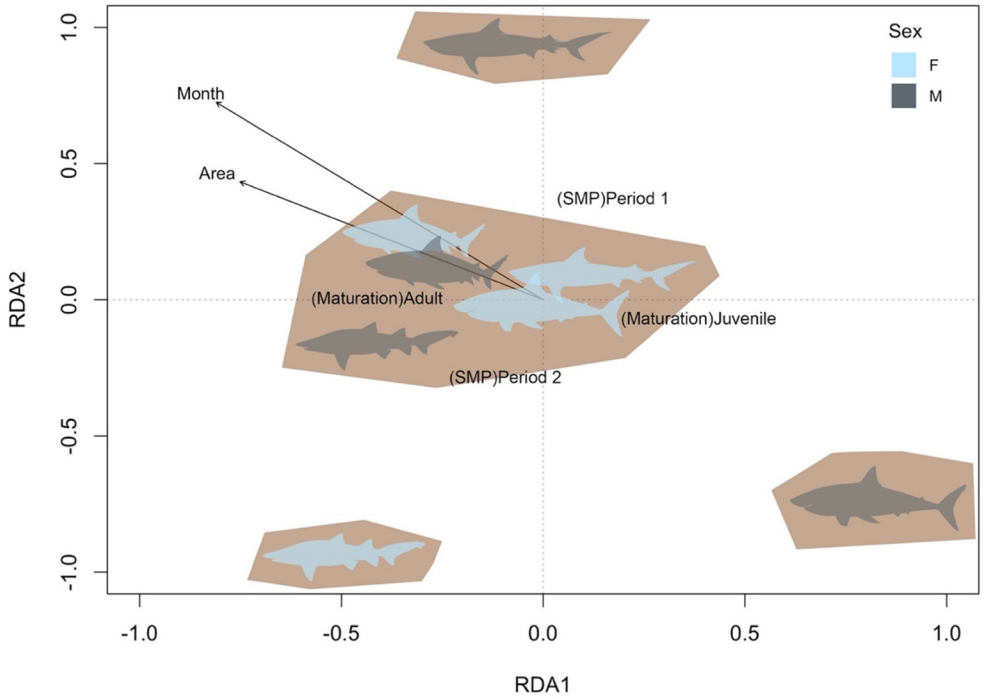


FIGURE 3 | RDA triplots (first two axes from the redundancy analysis RDA1 and RDA2) of the total catch of female and male (colour legend) grey nurse, white, tiger and bull sharks in the Shark Meshing Program (SMP) between 1998 and 2019 according to the significant groupings identified (shaded areas) and constrained by the corresponding significant variables maturation status (Maturation), month, SMP period (SMP) and netted area (Area). Arrows represent the positive increments for the continuous variables (month and area).

The nearshore occurrence of both threatened shark species increases during periods of water temperatures below 20°C (Lee et al., 2018), particularly in rocky reef and sandy bottoms areas. Our findings corroborate catches of grey nurse sharks in the South African bather protection program (Wintner and Kerwath, 2018), with increased catch apparently driven by cold water upwellings (Roughan and Middleton, 2002; Oke and Griffin, 2011), particularly in the north of the SMP area (Everett et al., 2014). Juvenile white sharks occupy this region as part of their nursery (Bruce and Bradford, 2008, 2012) suggesting increased catch may be related to their exploration of their nursery area. Grey nurse sharks migrate southwards during the spring-summer period (Otway and Ellis, 2011).

White sharks were more frequent in meshed areas with higher proportions of beach covered by the SMP nets (proportion of beach netted > 40%). This, combined with higher occurrence during productive cold-water periods, suggests that when following their prey into the shallows these sharks become more vulnerable to capture. Even though proportion of beach netted was not a significant variable for grey nurse sharks, the southern increase in captures occurred at beaches with a high proportion of the beach covered by the net (**Figure 1**) and removing these predicted a reduction of catch of ∼40% for the species (**Table 1**). Because SMP nets are set close to the beach, these sharks might have a higher likelihood of being caught at these shorter beaches as the reef is in proximity on either side of the net. Instead of being removed, these nets could be moved to deeper waters in an attempt to reduce captures of grey nurse sharks.

The southern region caught more tiger and white sharks, with white sharks mostly caught during the latter half of the season (January to April) coincident with known broadscale movements (Bruce et al., 2019), supporting South African research suggesting white shark catch was related to longshore movements (Cliff et al., 1989). The increased catch of both white sharks and grey nurse sharks in the nets on either extremity of the SMP region may therefore be due to the "boundary effect" first described by Cliff et al. (1989). The boundary effect proposes that sharks are more likely to be captured when they first encounter nets on the edge of these programs. Removing nets from extreme locations would not necessarily reduce shark catches but simply move the boundary. A more effective approach to reduce mortality of threatened species may rather be to change practises, for example deploying alternative mitigation strategies.

Addressing the Wicked Problem

Management of human-wildlife conflicts requires that both the human and ecological dimensions of the problem are properly identified and accounted for (Carter et al., 2014; Teixeira et al., 2021). Recognising local attitudes toward the animals involved is often among the main strategies to tackle these situations (Madden, 2008; Goodale et al., 2015; Dillon et al., 2016). This includes tolerance of wildlife presence (Bruskotter and Wilson,

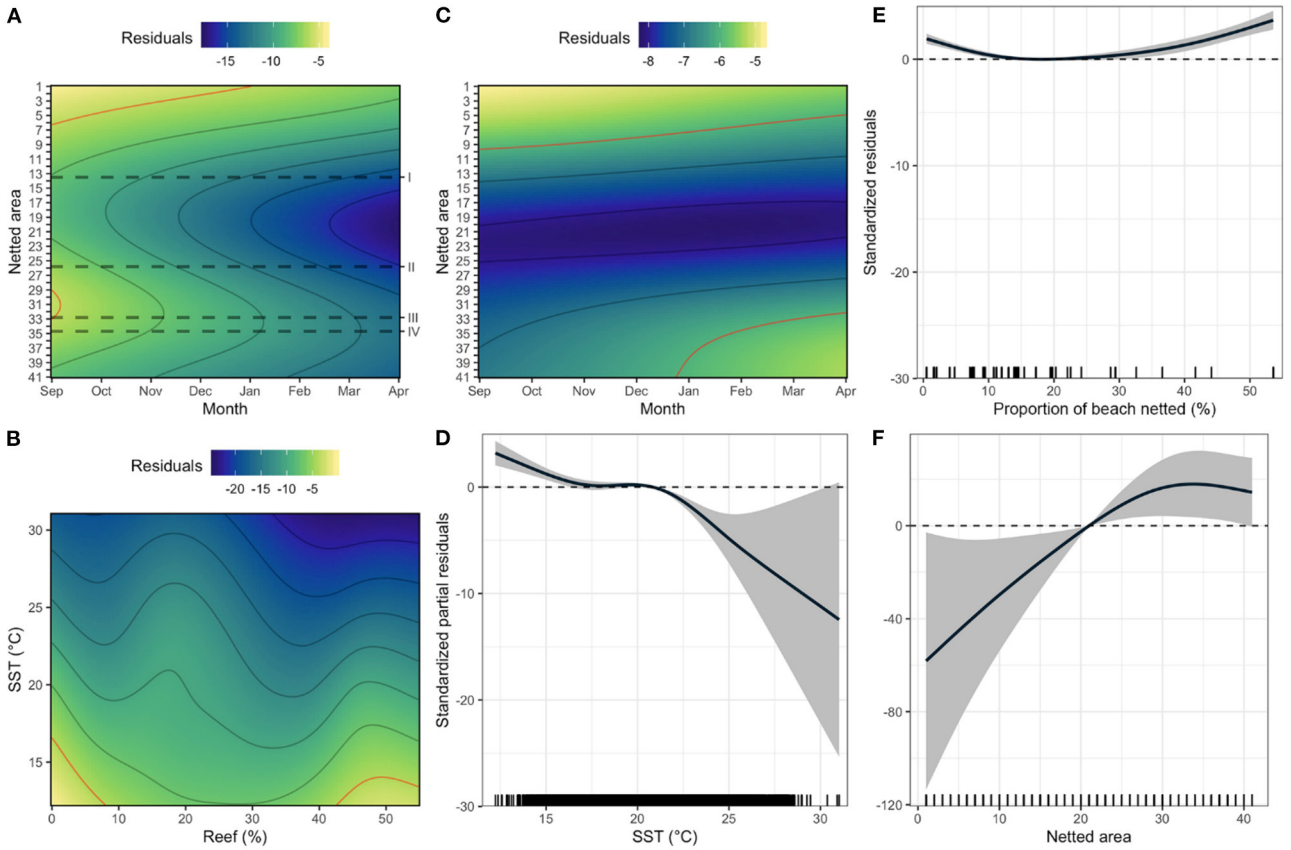


FIGURE 4 | Fine-scale zero-inflated generalised additive models of **(A,B)** grey nurse, **(C–E)** white, and **(F)** tiger shark occurrence in the Shark Meshing Program catches. Spatial-temporal effects of netted area and month **(A,C)**, and **(B)** sea surface temperature (SST) and percentage of reef area (Reef), and the individual effects of **(D)** SST, **(E)** proportion of beach netted, and **(F)** netted area. Colour scales, shaded areas and solid and dashed lines **(D–F)**, respectively, represent the respective model residuals, the 95% confidence intervals, the raw data and the null effects. The red isolines **(A–C)** represent the highest residual contours for the corresponding models. Dashed lines **(A)** depict the grey nurse shark aggregation sites (I = Foggies, II = Long Reef, III = Magic Point, IV = South Marley).

2014; Slagle and Bruskotter, 2019), which in turn is affected by personal experiences such as being involved in direct contacts or attacks by these animals (Treves and Bruskotter, 2014; Dorresteijn et al., 2016).

At one end of our matrix, maintaining the status quo of the SMP would result in ongoing catches of threatened grey nurse and white sharks, and zero impact on reducing this Key Threatening Process to their population viability. In Australia, there are strong pressures toward adapting bather protection programs, particularly in Western Australia and New South Wales (Fraser-Baxter and Medvecky, 2018), suggesting that this option would not be supported by NGOs nor beach goers. Simultaneously, this option has limited value to the government organisation charged with enhancing the conservation status of these species. However, despite public and NGOs being supportive of taking nets out of the water, complete removal of the SMP may exacerbate the wicked problem due to the possibility of increased risks to human safety. Therefore, we focus on the evidence supporting the replacement of nets with alternatives.

It is unlikely that all nets could be replaced by new methods as logistically replacing nets with SMART drumlines off beaches further than 15 km from a port is unfeasible if sharks are to survive (Tate et al., 2019). However, traditional drumlines could be installed at remote beaches as they do not require immediate response following a capture and incur lower mortality than nets (Sumpton et al., 2011). Survivorship in traditional drumlines off Queensland varies considerably (grey nurse = 50%, white = 47.5%, tiger = 31%, bull = 25.9%; Sumpton et al., 2011). They may represent a compromise by reducing mortalities of threatened species while still providing some mitigation (Cliff and Dudley, 2011), however they are generally not supported by the majority of the community (Gibbs and Warren, 2014).

The release of potentially dangerous sharks did not directly correlate with higher incidence of shark bites in South Africa (Cliff and Dudley, 2011), nor has it in NSW since it was formally implemented in 2009, supporting the use of non-lethal measures for mitigating shark hazard. Although approximately half of grey nurse sharks are released following capture in the SMP (Dalton and Peddemors, 2018), there

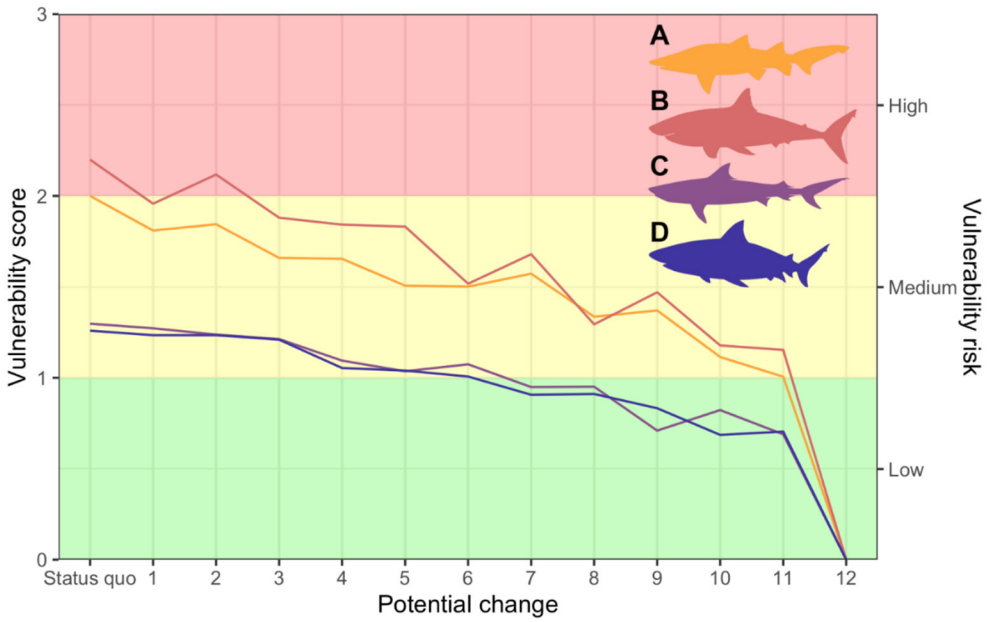


FIGURE 5 | Variation in vulnerability scores for each shark species (**A** = grey nurse; **B** = white; **C** = tiger; **D** = bull) as a function of the potential changes identified (**Table 2**). Background colours separate the total vulnerability score range into three groups according to the same "traffic light" pattern for vulnerability risk (green = low; yellow = medium; red = high) used for the wicked matrix (**Table 2**).

is little information on post-release survivorship. Grey nurse sharks are prone to stress-induced lactic acid build-up and subsequent mortality following capture (Smale et al., 2012; Otway, 2015). Apparently positive post-release survival from the SMP (Dalton and Peddemors, 2018) and the two trials of shark nets on the NSW North Coast (NSW DPI, 2017, 2018) suggests that SMART drumlines will lead to even higher post-release survivorship for this species. SMART drumlines do not appear to attract potentially dangerous sharks into nearshore areas, however, neither are they or any other existing mitigation measure capable of completely stopping sharks from using a particular area (Guyomard et al., 2020). Combining SMART drumlines with aerial surveillance (Simmons and Mehmet, 2018) could provide a potential alternative to shark nets.

Drones are a cost-effective substitute for manned aircraft, however, their effectiveness still depends on the ability of the pilot to identify a shark in variable environmental conditions (Butcher et al., 2019). The development of artificial intelligence is likely to reduce reliance on pilots in future, but there will also still be issues related to proximity with airports and other critical infrastructure. Sole reliance on aerial surveillance may, therefore, not be practical in many coastal regions. Drones have been shown to be capable of acting as aerial support for shark detection in nearshore waters (Butcher et al., 2019; Gorkin et al., 2020), however, until flight automation becomes more advanced it is unlikely to be considered viable at understaffed beaches for beach authorities. They could, however, be applied in combination with SMART drumlines off more popular beaches, e.g., Bondi Beach where high grey nurse shark and low white shark captures, coupled with the popularity of this Australian beach, point to a high likelihood of success.

Recognising the ecological patterns underlying a conservation issue while including knowledge of human behaviour is an effective strategy for solving wicked problems, but the science must be evidence-based (Game et al., 2014; Mason et al., 2018). Grey nurse sharks aggregate at particular locations along the east coast of Australia (Bansemer and Bennett, 2008) leading to calls to remove nets near aggregation sites (Barker and Williamson, 2010). However, this and previous studies (Green et al., 2009) found no evidence that proximity to aggregation sites had any influence on capture probability (**Supplementary Table 4**), implying that removal of these nets would be an ineffective strategy to reduce the impacts of the SMP, possibly misdirecting efforts. This argues strongly for evidence-based conservation decision making, particularly when sensitive to public scrutiny. It is important to emphasise that the nets are not a barrier that prevents sharks from interacting with beach goers as they do not cover the entire beach, rather they are a fishing agent. Research trials of shark nets on the NSW north coast found that only 6 out of 145 animals (4%) were caught on the shoreward side of the nets (NSW DPI, 2018). By contrast in South Africa, 35% of the sharks caught were done so on the shoreward side of the nets, i.e., when moving out to sea (Dudley, 1997). While removing nets might be considered controversial for risk-averse government agencies, it is evident from our findings that management efforts focused at only two beaches (i.e., Stockton and Bondi) could substantially improve shark conservation with little projected risk for people in this region, provided alternative mitigation measures are implemented in their place.

In human-wildlife interactions people are not only part of the problem but the essential component to finding effective solutions toward coexistence (Frank and Glikman, 2019). Many human-wildlife conflicts involve humans venturing into wildlife habitat, or exploiting areas in new ways, whether becoming exposed to sharks in the sea or large dangerous wildlife on land. In the case of shark bite risk, while management agencies are tasked with developing public safety strategies the public can also contribute to lowering their own risk, for example, by wearing personal electronic deterrents (Bradshaw et al., 2021). Balancing human risk with increasingly disturbed environments and ever diminishing populations of wildlife (Bar-On et al., 2018) requires robust approaches. Effectively measuring stakeholder attitudes toward potential alternatives for human-wildlife conflict can be challenging, as they may have divergent perspectives toward wildlife (Ceauşu et al., 2019), and these might further vary with time depending on people's personal experiences, culture and changes in beliefs. It is also important to weight the impacts and trade-offs involved in the risk of conflicts during recreational activities (Kubo and Shoji, 2014), including their decision making process around the threat of shark bites. Under the context of the SMP, no single definite solution could be found to replace current methods as some stakeholders, including members of the public, were found to agree with the use of shark nets (Gray and Gray, 2017). Any changes to the program are likely to be location-specific, depending on feasibility of use for each of the alternatives proposed in this study, whilst simultaneously accounting for the attitudes of the people involved. While we acknowledge that our findings might not be definitive, we hope they will provide decision makers with sufficient understanding to move toward a more holistic approach to managing marine ecosystems in which humans and sharks can coexist. The framework adopted in our study incorporates (i) identification of species-specific occurrence patterns, (ii) determining potential actions toward reducing impacts for wildlife while keeping people safe, (iii) evaluation of stakeholder attitudes toward proposed alternatives. Our results indicate that integrating this framework into human-wildlife conflict resolution strategies is not only applicable to other programs of mitigating shark bite risk, but also for managing human-wildlife conflicts in other situations where empirical data on species occurrence and ecology are available.

DATA AVAILABILITY STATEMENT

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found at: https://www.sharksmart.nsw.gov.au/__data/assets/pdf_file/0009/1246275/smp-2019-2020-annual-performance-report.pdf.

ETHICS STATEMENT

Ethical review and approval was not required for the animal study because data was obtained by a long-running (since 1937) government program.

AUTHOR CONTRIBUTIONS

YN, VP, AS, and RH conceived the ideas and designed the methods. VP, AS, and MG collected the data. YN analysed the data and led the writing of the manuscript. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.720741/full#supplementary-material

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COMMUNITY CASE STUDY
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Developing an Evidence-Based Coexistence Strategy to Promote Human and Wildlife Health in a Biodiverse Agroforest Landscape

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Agroforest mosaics represent one of the most extensive human-impacted terrestrial systems worldwide and play an increasingly critical role in wildlife conservation. In such dynamic shared landscapes, coexistence can be compromised if people view wildlife as a source of infectious disease. A cross-disciplinary One Health knowledge base can help to identify evolving proponents and threats to sustainable coexistence and establish long-term project goals. Building on an existing knowledge base of human–wildlife interactions at Cantanhez National Park (NP), Guinea-Bissau, we developed a causal pathway Theory-of-Change approach in response to a newly identified disease threat of leprosy in the Critically Endangered western chimpanzee (*Pan troglodytes verus*). The goals of our project are to improve knowledge and surveillance of leprosy in humans and wildlife and increase capacity to manage human–wildlife interactions. We describe the core project activities that aim to (1) quantify space use by chimpanzees across Cantanhez NP and determine the distribution of leprosy in chimpanzees; (2) understand the health system and local perceptions of disease; and (3) identify fine-scale risk sites through participatory mapping of resources shared by humans and chimpanzees across target villages. We discuss the development of a biodiversity and health monitoring programme, an evidence-based One Health campaign, and a One Health environmental management plan that incorporates the sharing of space and resources, and the disease implications of human–non-human great ape interactions. We demonstrate the importance of multi-stakeholder engagement, and the development of strategy that fully considers interactions between people, wildlife, and the environment.

Keywords: human–wildlife coexistence, infectious disease, leprosy, One Health, great apes, West Africa, theory of change

INTRODUCTION

With continued human population growth and the associated expansion of human activities, an estimated 95% of Earth's surface has been modified by humans (Kennedy et al., 2020). There is growing recognition that landscapes shared by humans and wildlife, including agroforest mosaics with varying intensity of human activity, will play an increasingly critical role in the conservation of threatened species (Ellis and Ramankutty, 2008; McKinney, 2015). This is of particular importance in the tropics that harbour ∼80% of species worldwide and are undergoing rapid environmental change (Döbert et al., 2014; Hockings and McLennan, 2019). In shared landscapes, the urgent need to conserve species that are hovering on the edge of extinction demands the coexistence of people and wildlife, and this poses one of the greatest conservation challenges of the twenty-first century (McLennan et al., 2017; Frank et al., 2019).

Understanding how humans and wildlife use shared landscapes, the nature and implications of their interactions, and how these vary over time, is crucial for developing conservation strategies. Human–wildlife coexistence does not imply the absence of (competitive) interactions or risks, including bi-directional aggression or disease transmission, but negative interactions must remain at a tolerable level to both people and wildlife (Carter and Linnell, 2016). To this end, greater understanding and consideration of the human ecological and socio-political dimension alongside that of wildlife biology are paramount, with conservation strategies ultimately needing to improve the lives of local people (Adams et al., 2004; Dore et al., 2017; Hill et al., 2017; König et al., 2020; Pooley et al., 2020).

Many threatened species are found in tropical countries where a disproportionate amount of people live in poverty. Poverty is not only economic; it encompasses a range of diverse issues including lack of access to healthcare, education, and clean water sources. Conservation programmes that promote coexistence can help to alleviate poverty in rural communities within or adjacent to wildlife habitats. This can be achieved through direct approaches such as helping to secure increased income for rural communities and improving access to healthcare, and indirect approaches such as safeguarding traditional rights, cultural values, ecosystem services, or capacity building of local groups and institutions (United Nations, 2021). For this to succeed, conservationists must work with diverse stakeholders to ensure that conservation approaches are built within a framework that actively incorporates social equity, as well as poverty alleviation and well-being among local communities. Of considerable importance is developing strong working relationships and collaborations amongst local stakeholders such as user groups, youth, student and women's associations, as well as national and international institutions.

INFECTIOUS DISEASE AND ONE HEALTH

The risk of disease exchange between humans and wildlife means that coexistence can be compromised when people view wildlife as a source of infectious disease and resort to hostile actions (Bicca-Marques and de Freitas, 2010; López-Baucells et al., 2018; MacFarlane and Rocha, 2020). Infectious disease emergences or re-emergences (i.e., that have either recently appeared or were already present, and which are increasing in incidence or geographic range) are predicted to become more frequent in both humans and wildlife as a result of anthropogenic and environmental changes. Consequently, the potential for cross-species disease transmission to create serious conservation problems, on top of the obvious risks to public health, are also expected to increase. For wildlife, beyond the threats of retaliatory killings by people, there are risks of infection-induced mortality that can arise in two ways. Firstly, zoonotic diseases that are endemic to wildlife and can transmit to humans may become problematic if concurrent threats (e.g., habitat loss by increasing densities or stress) exacerbate infection prevalence and/or severity and lead to disease re-emergence. Such re-emergence could imperil both the animal reservoir species, as well as humans, particularly as habitat destruction, road building and hunting are forcing wildlife into shifting their distributions to utilise human-impacted habitats. Indeed, zoonotic pathogens are thought to have given rise to >60% of known human pathogens, and 75% of emerging infectious diseases in humans to date have a zoonotic origin (Taylor et al., 2001). Secondly, infectious diseases of human origin can emerge in wildlife through spill-overs or host jumps when there is sufficient contact between humans and wildlife. For example, the transmission of metapneumovirus from humans to wild mountain gorillas (*Gorilla beringei beringei*) was shown to be responsible for the death of gorillas in Rwanda in 2011 (Palacios et al., 2011). Because successful pathogen establishment is made more likely by the close phylogenetic proximity between humans and non-human primates (hereafter primates), any increase in human-primate interactions will intensify bidirectional risks of disease exchange (Gillespie et al., 2008). Addressing the issue of disease exchange between humans and wildlife requires adopting a One Health approach which affirms the interconnectedness between people, animals and the environment.

RESPONDING TO A NEWLY IDENTIFIED DISEASE THREAT IN CRITICALLY ENDANGERED WESTERN CHIMPANZEES AT CANTANHEZ NP, GUINEA-BISSAU

Cantanhez NP (1,067 km^2), Guinea-Bissau is inhabited by over 24,000 people (**Figure 1**). Cantanhez NP is characterised by a mosaic of coastal sub-humid forest patches, mangroves, savannah grassland, woodland and agriculture. Since 2013, the Cantanhez Chimpanzee Project has been building a knowledge base of human–wildlife interactions. In 2015, we first observed symptoms of leprosy (*Mycobacterium leprae*) in chimpanzees at Cantanhez NP, with molecular confirmation of *M. leprae* as the causative agent in 2018 (Hockings et al., 2021). This is the first evidence of leprosy in wild nonhuman great apes and in any wildlife in Africa.

Leprosy is a neglected tropical disease, with ∼210,000 new human cases reported every year, of which 2.3% are located in West Africa (WHO, 2021a). Leprosy has a long incubation

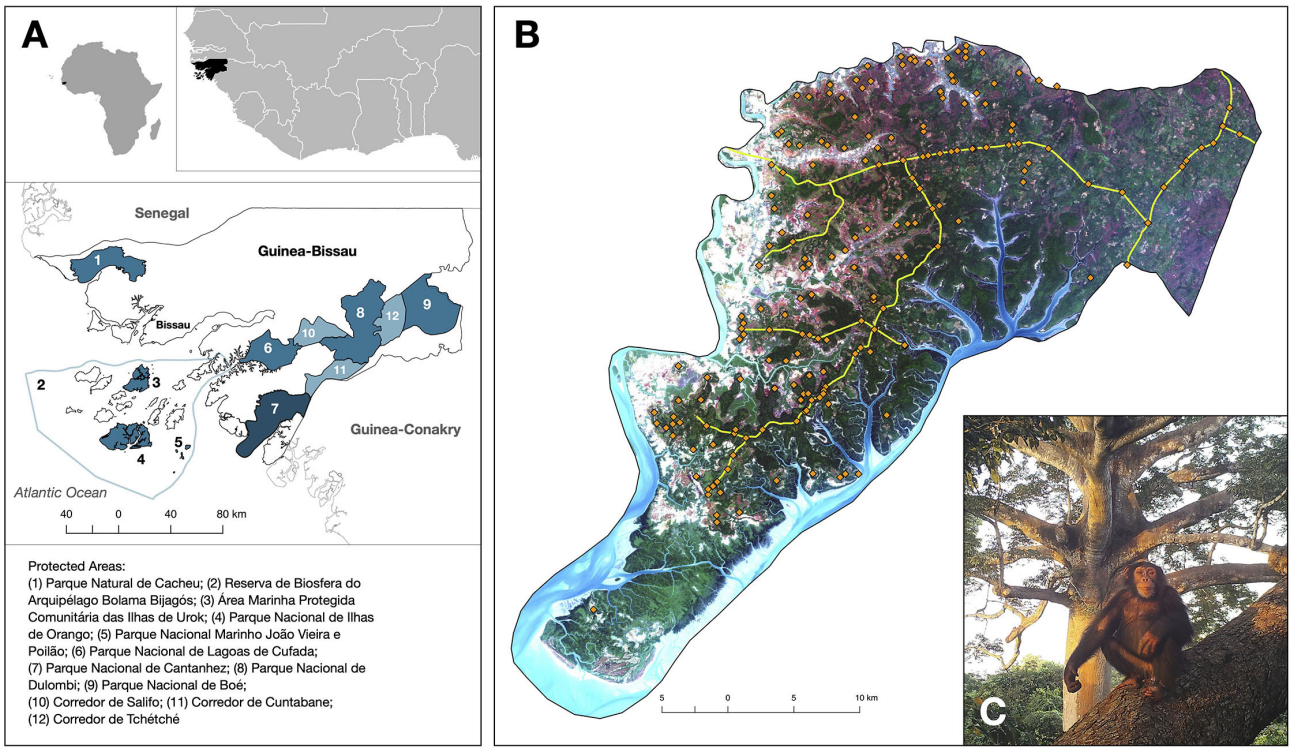


FIGURE 1 | (A) Guinea-Bissau (36,125 km^2) hosts a network of marine and terrestrial Protected Areas, including Cantanhez National Park shown in dark blue; **(B)** Cantanhez National Park roughly corresponds to the Bedanda Sector (24,293 inhabitants in 2009), administrative designation for the Cubucaré peninsula, located in the Tombali Region. Cantanhez National Park includes a road network and about 200 villages and settlements of varying sizes, from about 10 to over 1,000 inhabitants; **(C)** the Critically Endangered western chimpanzee is a national flagship species in Guinea-Bissau. Cantanhez National Park hosts ∼10 chimpanzee communities across protected forest fragments. The map of Cantanhez National Park contains modified Copernicus Sentinel data from 2017 processed by Sentinel Hub.

time of several months to 30 years in humans, with an average of 5 years. Untreated infections result in permanent damage, including an inability to feel pain that leads to a characteristic loss of extremities from repeated injury or infections. Transmission is traditionally believed to occur primarily via aerosolised nasal secretions and entry through nasal or respiratory mucosae, and is therefore considered most common between individuals in close contact (Lastória and Abreu, 2014; Araujo et al., 2016). However, the role of other transmission routes is unclear. *Mycobacterium leprae* can circulate in other animal hosts in the wild, such as the nine-banded armadillos (*Dasypus novemcinctus*) in South and North America and red squirrels (*Sciurus vulgaris*) in the United Kingdom, with the pathogen likely transmitted from humans centuries ago (Hamilton et al., 2008; Sharma et al., 2013; Avanzi et al., 2016). Our recent study on leprosy circulating in wild chimpanzees also suggests environmental reservoirs as potential sources of infection (Hockings et al., 2021).

In response to the detection of leprosy in chimpanzees, and concern that it might result in conflicts with local people, we built on our existing knowledge base to develop a collaborative, cross-disciplinary One Health project. In this paper, we first provide broad context on the social, historical, and biodiversity-rich landscape at Cantanhez NP as a foundation for understanding human–wildlife coexistence. We then present a causal model to promote public health and conservation and detail project activities, including developing a biodiversity and health monitoring programme; understanding the healthcare structure and its use by the local population; evaluating local knowledge of and perception of disease; and mapping human–wildlife interactions and risk hotspots. We discuss how these activities feed into project outputs to ensure improved knowledge and surveillance of leprosy in humans and wildlife, with enhanced capacity to manage human–wildlife interactions across the landscape. The research involving wildlife and human participants was reviewed and approved by the Ethics Committee of the University of Exeter (Refs: eCORN002520 v3.1 and eCORN002528 v3.1), and human participants provided their informed consent to participate in this study.

UNDERSTANDING HUMAN–WILDLIFE COEXISTENCE IN CANTANHEZ NP

The Social and Historical Landscape of the Cubucaré Peninsula

Guinea-Bissau is a West African country with a diverse ethnic, cultural and religious history. Cantanhez NP is inhabited mostly

by Balanta people, as well as the Nalu, Fulani and Sussu, among many others (Catarino and Palminha, 2018). Over the past decades, populations from different ethnic groups have been moving to this region, mainly from the Republic of Guinea (Temudo and Abrantes, 2014; Parathian et al., 2018). Land use at Cantanhez has changed over time, with the flooded rice culture developed by the Balanta now widely practised (Temudo, 2009). Small-scale production of mango and citrus trees once allowed the natural vegetation to be maintained, with limited deforestation (Temudo, 2009), but from the 1980s onwards the installation of cash fruit crops, including cashew, transformed the landscape for people and wildlife (Temudo and Abrantes, 2014; Havik et al., 2018). With the widescale introduction of fruit crops, frugivores including the chimpanzee, have gradually incorporated these new food sources into their diets, changing the nature of their interactions with people (McLennan and Hockings, 2014; Bessa et al., 2015).

In symbolic and political terms, the inhabitants of Cantanhez NP are divided between the "owners," the Nalu, founders of the territory, and "guests," the population of other ethnic groups to whom the Nalu have given permission to settle there. The Nalu have an intricate relationship with the landscape and wildlife. According to Nalu' pre-Islamic ontology, all beings including supernatural entities, humans, animals and plants, form a system of relations (Frazão-Moreira, 2009, 2015). Within the Nalu territory there are sacred zones that are commonly referred by the Creole term "matu sagrado" (sacred forest). These forests are a landmark of ecological, political and cultural history. The underlying components of a pre-Islamic ontology combined with Muslim beliefs are key to understanding human–chimpanzee coexistence (Sousa and Frazão-Moreira, 2010; Costa et al., 2017). Nalu people have the general belief that all non-human species have reputed access to resources in ancestral lands and the Muslim aram prevents the killing and eating of any animal with canine teeth, including primates. The Nalu believe "dari i pekador" (the chimpanzee is human) and acknowledge the physical and behavioural similarities that chimpanzees and humans share (Sousa and Frazão-Moreira, 2010). Animist ontology guides local beliefs that non-human species exist either as true animals, or some other animal form transformed by "irãs" (supernatural beings), with humans and chimpanzees being able to shape-shift into each other's physical forms (Sousa et al., 2017, 2018). Acts perceived negatively by people (such as unprovoked attacks on local persons by chimpanzees) are sometimes attributed to people taking the form of chimpanzees.

In the 1990s, on the initiative of the Non-Government Organisations Acção para o Desenvolvimento, Tiniguena and Alternag, with the support of International Union for Conservation of Nature (IUCN), a project was set up to conserve the remaining forests in the Tombali region, which in 2011 led to the creation of the Cantanhez NP (**Figure 1B**). Since 2011, Cantanhez NP has been managed by the Institute for Biodiversity and Protected Areas (IBAP) following an IUCN Category V protected area, acknowledging that "the interaction of people and nature over time has produced an area of distinct character with significant ecological, biological, cultural and scenic value: and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values" (IUCN, 2016). The local management committee (includes 35 people from 14 villages) actively participates in the management of Cantanhez NP which is divided into three zones, including the core zones, largely consisting of protected coastal forest blocks where hunting and logging activities are forbidden, buffer zones, where hunting by local residents is allowed but felling of large trees by residents is subject to permission from IBAP, and agricultural zones (Catarino and Palminha, 2018; IBAP, 2020).

The Environment, Biodiversity and Human–Wildlife Resource Overlap

Guinea-Bissau lies within the Guinean forest-savannah mosaics, a biodiverse ecoregion buffering the Guinean moist forests in the south and the West Sudanian savannah in the north (**Figure 1A**). The climate in Guinea-Bissau is characterised by a rainy season from mid-May to the end of October and a long dry season from November to mid-May. Cantanhez NP hosts a wealth of wildlife species. Six diurnal primate species are present in Cantanhez NP, including the western chimpanzee (**Figure 1C**), Temminck's red colobus (*Piliocolobus badius temminckii*), king colobus (*Colobus polykomos*), Guinea baboon (*Papio papio*), Campbell's monkey (*Cercopithecus campbelli*) and green monkey (*Chlorocebus sabaeus*). Cantanhez NP is also home to numerous ungulate species including buffalo (*Syncerus caffer*) and bushbuck (*Tragelaphus scriptus*), and biomonitoring efforts have recently confirmed the presence of rare and elusive species, including the African golden cat (*Caracal aurata*), giant pangolin (*Smutsia gigantea*) and in the North of the park, the African elephant (*Loxodonta africana*) (**Supplementary Table 1**).

As a large-bodied and socio-ecologically flexible species, chimpanzees are able to adapt to complex and dynamic environments, including human-impacted landscapes if they are not hunted (Hockings et al., 2015; Bersacola et al., 2021). At Cantanhez NP, humans and chimpanzees show substantial overlap in their use of space, and encounter each other regularly on roads, paths and around the edges of villages and agricultural areas. Humans and chimpanzees use at least 27 of the same wild fruit species, including oil palm (*Elaeis guineensis*), velvet tamarind (*Dialium guineensis*), and saba (*Saba senegalensis*) (Hockings et al., 2020), with chimpanzee use of space driven partly by the availability of oil-palm fruit (Bersacola et al., 2021). Although the Caiquene-Cadique community of chimpanzees in central Cantanhez NP use areas away from villages and agriculture more intensively, they optimise their foraging strategies when wild fruits are scarce by increasing their use of village areas with cultivated fruits (Bersacola et al., 2021). Despite beliefs that prevent people from hunting chimpanzees and consuming their meat, negative interactions between people and chimpanzees can occur over highly valued crops such as oranges and papaya, with chimpanzees chased and shot at by local people to keep them away from crops during fruiting seasons (Sousa and Frazão-Moreira, 2010). People are more tolerant of chimpanzees foraging on cashew pseudofruit, as the commercially valuable cashew nut remains unharmed. Farmers

perceive that the chimpanzees leave cashew nuts in more manageable piles making harvesting easier (Hockings and Sousa, 2012).

A COLLABORATIVE PLANNING APPROACH TO IMPROVE PUBLIC HEALTH AND PROMOTE BIODIVERSITY CONSERVATION

The development of strategy to help protect threatened species and alleviate poverty in rural communities within or adjacent to wildlife habitats can be challenging. We used a Theory of Change causal model (i.e., an outcomes-based approach to design, implement and evaluate initiatives) to encourage critical thinking and demonstrate linkages between the complex network of factors to promote public health in this biodiverse agroforest landscape (**Figure 2**). A fundamental part of our project is the consolidation of a multi-stakeholder, cross-disciplinary, multi-institutional collaborative approach to promote conservation and human health in Cantanhez NP.

BIODIVERSITY AND HEALTH MONITORING PROGRAMME

Measuring Chimpanzee Distribution and Intensity of Space Use

As part of the Biodiversity and Health Monitoring Programme (BHMP), we established transect routes and camera trap sampling points across Cantanhez NP in December 2019 (**Supplementary Figure 1**). Transects and camera traps covered habitat types across the landscape mosaic, including secondary forest, woodland, mangroves, orchards, shifting cultivation fields, villages, roads and remnant forest strips dominated by oil palms. We carried out monthly transect surveys using Cybertracker and checked camera traps monthly. We conducted distance sampling along line transects and recorded chimpanzee nests, direct encounters with wildlife, and animal and human signs, following standardised protocols (Buckland et al., 2015).

As of March 2021, biomonitoring surveys consisted of 400 km of transect survey effort and 11,884 camera trap days across 13 protected forest blocks and surrounding agro-forest mosaics. **Supplementary Table 1** shows the confirmed presence of all wildlife species recorded during surveys. Chimpanzees are distributed across the park, with nests recorded in 44% of the transects. Nests were observed more frequently across the central part of Cantanhez NP (86%, of total N of nest observations = 72), where sub-humid forest cover is more extensive compared to the drier savannah-woodland-forest mosaic in the northern part. We analysed camera trap data collected during the first biomonitoring season between January and July 2020 (7,514 camera trap days). We extracted 901 independent events for chimpanzees, defined as images or sequences of images at least 60 mins apart from the previous images of the same species at the same location. We employed Bayesian spatial modelling and extracted a prediction map for intensity of space use by chimpanzees across Cantanhez NP (**Figure 3A**, see figure caption for more details on the analysis).

Monitoring Leprosy in Chimpanzees

We first observed clinical signs of leprosy in chimpanzees through the analysis of camera trap data in 2015. We then conducted molecular analyses of chimpanzee faecal samples and confirmed *M. leprae* as the causative agent of the lesions observed in chimpanzees (see Hockings et al., 2021, for details). From 2015 to 2021, we were able to determine the occurrence of advanced leprosy (multibacillary) in five chimpanzee communities across the park, with *M. leprae* likely transmitted between individuals within this population. We identified seven affected individuals (three adult females, two adult males and two adults of unconfirmed sex), and possible cases of leprosy in three other individuals. Advanced leprosy symptoms in chimpanzees include hair loss, plaques and nodules that cover different areas of their body (limbs, trunk and genitals), facial skin hypopigmentation, facial disfigurement as well as ulcerated and deformed hands (claw hand) and feet (Hockings et al., 2021) (**Figure 3B**). Paucibacillary cases in chimpanzees, where bacterial levels are low, can easily go undetected as minor physical manifestations of leprosy such as localised hair loss are challenging to observe on camera trap footage. Data collection using additional camera traps in affected chimpanzee communities, as well as faecal sampling for determining the prevalence of leprosy in chimpanzees are ongoing. These long-term data allow us to determine changes in physical symptoms displayed by individuals over time and any changes in community prevalence of the disease.

HUMAN DIMENSIONS OF HEALTH

The Health System in Guinea-Bissau

Guinea-Bissau is strongly dependent on agriculture and fisheries and is among the poorest countries on Earth, with 67% population living below $1.90 USD/day. The health sector in Guinea-Bissau suffers from a lack of financial and material resources, as well as a lack of specialised human resources (Guerreiro et al., 2017, 2018a,b; Russo et al., 2017; Guerreiro, 2019). The national health system is organised into three levels: central, regional and local (**Supplementary Table 2**). In 2017, 66% of the population did not have geographic access to services, i.e., they live beyond 5 km from the nearest primary health care delivery structure (Guerreiro et al., 2017). According to the latest available data, in 2016, the number of doctors stood at 1.3 per 10,000 people, and the number of nurses and midwifery personnel at 5.9 per 10,000 people (WHO, 2021b). At regional and local levels, there are also some health facilities under the responsibility of non-governmental, religious or other organisations.

Leprosy Strategies and Treatment

The government of Guinea-Bissau has established sectoral action plans, following World Health Organization (WHO) guidelines. These include the Master Plan for the Control of Neglected Tropical Diseases in Guinea Bissau, which takes

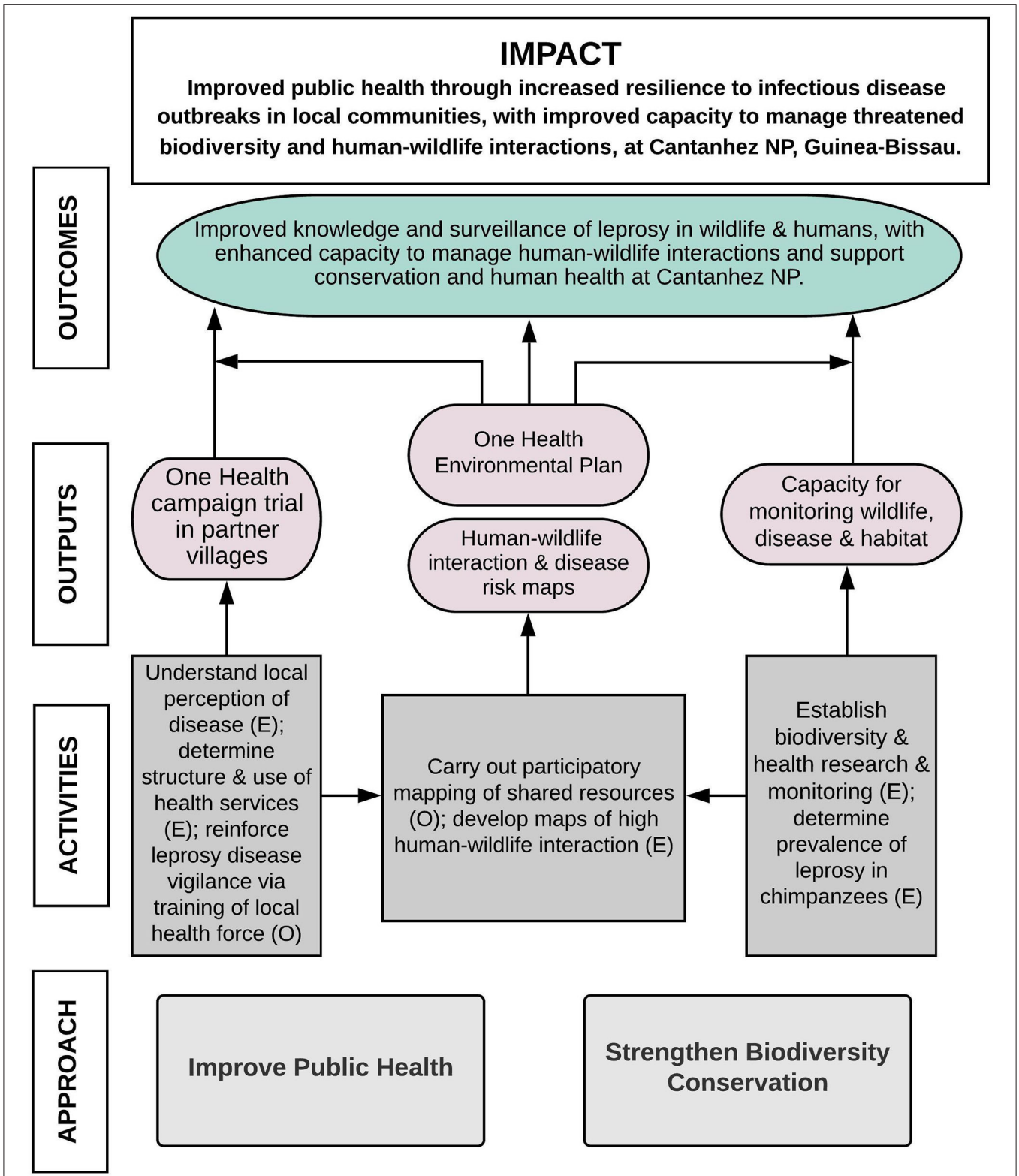


FIGURE 2 | A theory of change model including the project's: *Impact*—higher level objective that the project will help towards achieving; *Outcomes*—changes expected from the project and who is expected to benefit; *Outputs*—specific, direct deliverables of the project; and *Activities*—main tasks that the project will carry out. (E) represents executed activities and (O) shows ongoing activities. Our project Measurable Indicators include: (Outcome) By end of project, institutional capacity

(Continued)

FIGURE 2 | to monitor disease in wildlife in Cantanhez NP is increased to 80% compared to baseline (zero) through the implementation of biodiversity monitoring and the establishment of an IBAP surveillance team for sample collection and management of animal carcasses; (Output 1—Health Campaign) By the end of Year 3, at least 50% of campaign participants ($N = 30$ out of 60, including at least 50% women) demonstrate increased understanding in the links between environmental, animal and human health via One Health trial campaign in six partner villages compared to baseline pre-campaign; (Output 2—BHMP) By Year 2, the wildlife monitoring capacity in Cantanhez NP is increased to 80% compared to baseline 5% (based on number of park staff trained to record and analyse data, and existing training manuals); (Output 3—One Health plan) By the middle of Year 3, key wildlife habitat including corridors, and areas of high human–wildlife interaction and potential disease transmission are identified, and; by the end of Year 3, the plan is developed with stakeholders (Management Committee and other group representatives including from the Cantanhez Women's Associations) comprising at least 50% women; (Output 4—Leprosy response plan) By the end of Year 3, institutional capacity to sample, handle and dispose wild animal carcasses securely is increased by 100% compared to baseline zero; By Year 4, institutional knowledge to manage and respond to conflicts over leprosy disease (including mistrust of health services, retaliatory behaviour towards animals) and leprosy in humans is increased via the production of the first multi-stakeholder leprosy response plan in Guinea-Bissau. This project includes different health and conservation partner Institutions and Organisations in Guinea-Bissau, including the Institute for Biodiversity and Protected Areas, (IBAP), the Associação Nacional para o Desenvolvimento Local e Urbano (NADEL), and in Europe, including the University of Exeter in the United Kingdom, the Centre for Research in Anthropology in Portugal, and the Robert Koch Institute in Germany. Local partners include the women's groups and the local management committee, which is actively involved in the management of the park alongside IBAP.

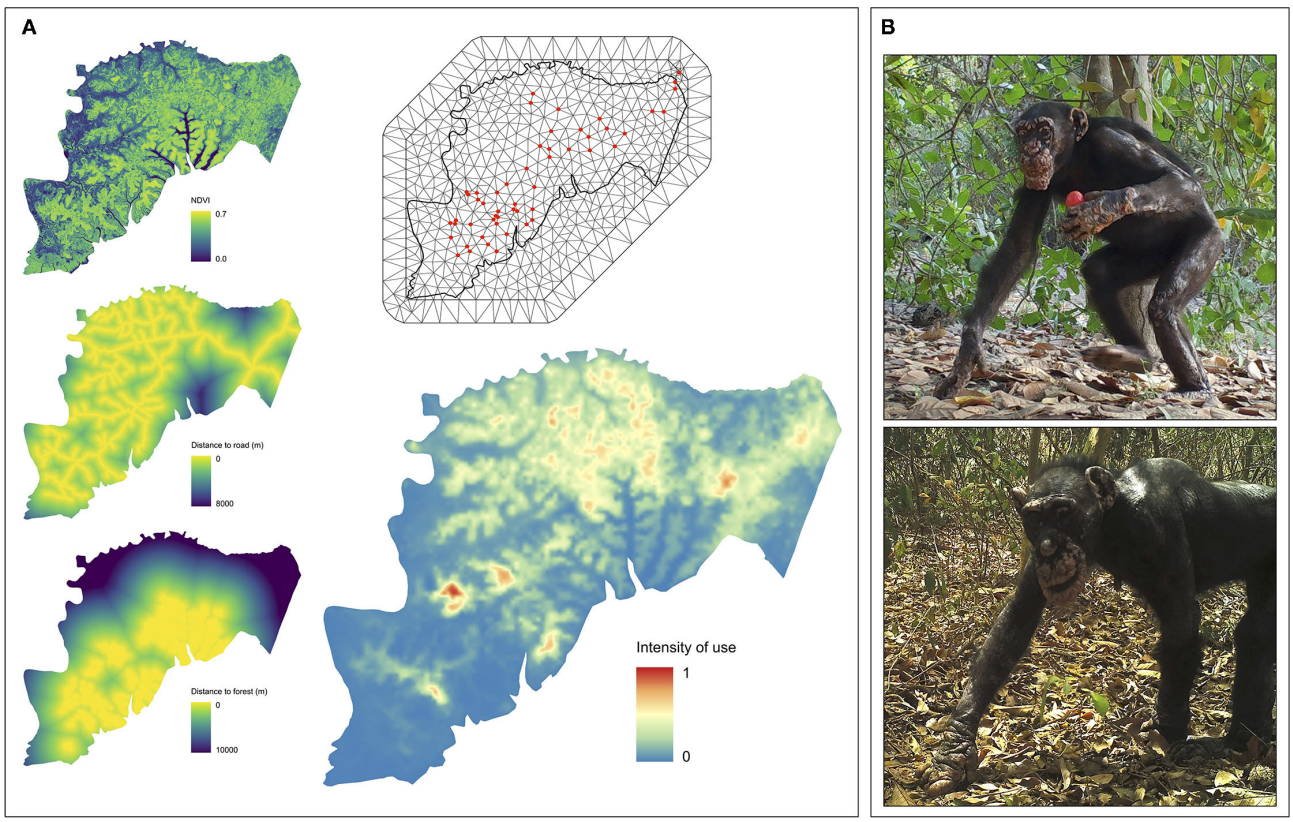


FIGURE 3 | Summary of results based on camera trap data from the first season of the biodiversity monitoring programme in Cantanhez NP. **(A)** Camera trap data were analysed to determine intensity of space use by chimpanzees. We employed Bayesian spatial modelling using the Integrated Nested Laplace Approximation (INLA) with Stochastic Partial Differential Equation (SPDE) approach (Lindgren et al., 2011; Rue et al., 2009) using the R-INLA package on R (Blangiardo et al., 2013; Blangiardo and Cameletti, 2015; R Core Team, 2020). Since its recent inception, this efficient statistical framework has become established across various scientific disciplines (Rue et al., 2017; Bakka et al., 2018) including wildlife ecology (e.g., Lezama-Ochoa et al., 2020). Its applicability to camera trap-based ecological research has also been recently demonstrated (Bersacola et al., 2021). Our model included seven covariates, including normalised difference vegetation index (NDVI), calculated from Sentinel-2 imagery dated 29 February 2020, as well as linear distance to the nearest protected forest block, distance to the nearest village, distance to large roads (newly improved 10 m-wide gravel roads) and distance to all roads (including small rural roads). Latitude and longitude were also included as covariates. This modelling framework uses a triangulation mesh of the study area to estimate the spatial autocorrelation amongst the sampling points and allows for the estimation of geostatistical data across a continuous field (interpolation). Bottom right shows the output map of the selected Binomial-SPDE model for chimpanzees, with the response variable constituting intensity of space use by chimpanzees at Cantanhez NP, represented by the sum of the number of occasions with positive detection (y) relative to the total number of sampling occasions (number of trials). **(B)** Two of the identified advanced leprosy cases in chimpanzees in Cantanhez NP, from top to bottom showing "Rita", an adult female from the Caiquene-Cadique community diagnosed with advanced leprosy based on camera trap footage and confirmed through molecular analysis in her chimpanzee community, and "Cristina", an adult female from the Faro Sadjuma community, with her diagnosis based on physical symptoms.

into consideration 13 priority diseases in Africa, including leprosy, and the National Tuberculosis and Leprosy Control Programme (MINSAP, 2014). Leprosy, according to the criteria of the WHO, is considered to be eliminated at the national level in Guinea-Bissau, because the number of patients on multidrug therapy is <1 case per 10,000 inhabitants (a value considered as the elimination target by the WHO). Nevertheless, leprosy is endemic with official figures showing between 50 and 70 new cases of leprosy detected each year (MINSAP, 2014). Importantly, not all cases of leprosy are identified; in its early stages the disease is easily confused with other conditions.

Cumura Hospital, located near Bissau, is the only health structure dedicated to the treatment of leprosy in Guinea-Bissau. It was founded in 1952 with the support of Catholic missionaries (particularly Italian Franciscan missionaries) to isolate leprosy patients, according to the medical knowledge of the time and the stigmatisation of the disease (Costa, 2010). Nowadays, the Cumura Hospital is managed by the Catholic Mission of Cumura and funded by the NGO AIFO (Amici di Raoul Follereau), following international scientific procedures concerning leprosy. Patients are mainly treated on an outpatient basis, although Cumura remains a permanent residence for some leprosy patients with disabilities. Beyond hospital work, the Cumura team has trained some nurses and community health agents. Cumura Hospital has treated patients from the Tombali region, although we do not currently have reliable counts of people with leprosy in Cantanhez NP. The population of Cantanhez NP locally receives support from "Type C" health centres (**Supplementary Table 2**), as well as from health facilities belonging to religious missionaries. If a patient with suspected leprosy is identified by a community health agent, the agent directs the patient to a "Type C" health centre. If treatment cannot be carried out there, the patient is directed to a "Type B" health centre at Cacine or Quebo, or to the Catió Regional Hospital. If a suspected leprosy case is identified, the patient should expect to be referred to Cumura Hospital.

Traditional medicinal systems are also present in Guinea-Bissau and are particularly important in contexts where the public health system is lacking or inefficient. In these systems, medicine cannot be dissociated from ontology and religious dimensions. In Cantanhez NP, there are various types of traditional healers: *curandeiros* (healers who can be from any ethnic group with different religions), *djambakus* (healers that perform divination and ritualised treatments based on traditional and pre-islamic Nalu beliefs), and *marabouts* (Muslim medicine-men). Healers have a refined knowledge of medicinal plants and treatments (Frazão-Moreira, 2009, 2016; Catarino et al., 2016). Patients consult these healers, seeking explanations for their health problem and its cure, either before going to the health centre or at the same time as consultations with nurses or doctors. Here, as in other parts of Africa, treatment itineraries are complementary and simultaneous, and are not culturally perceived as contradictory (Augé, 1984; Janzen and Green, 2003; Ribera, 2007).

LOCAL KNOWLEDGE OF LEPROSY AND ONE HEALTH

We conducted semi-structured questionnaires (Bernard, 1995; Kvale, 1996; Weller, 2015) with 92 participants including residents, healers and health workers across Cantanhez NP to identify local knowledge on leprosy transmission, diagnosis, and treatment (**Figure 4**, see figure caption for more details on methods). To help inform the development of a One-Health strategy, we conducted interviews with another 50 residents to understand perceptions and mitigation of risks of infectious disease transmission and barriers when accessing healthcare. In Cantanhez NP, leprosy is known by local people, but knowledge of symptoms, causes and transmission is limited, particularly at its early stage (**Figure 4A**). There was a lack of knowledge by both healers and health workers on how to treat leprosy and which health facilities to direct suspected leprosy patients, including no knowledge of the specialist leprosy hospital, Cumura. Zoonotic disease transmission is an issue that people show little awareness of, although there is some knowledge of factors that might increase the risk of zoonotic disease transmission including the consumption of fruit that an infected animal has touched, the consumption of meat from a sick animal, and the use of shared water sources. Many people, but not all, perceive wildlife carcasses as possible disease risks. To avoid disease transmission, people suggest maintaining a distance and not touching the carcass. However, perceptions of what to do in the event of encountering a sick wild animal vary (**Figure 4B**). Most people said they would go to a hospital or a health centre if they become ill, with some also visiting healers. However, people experience difficulties (from a lack of finances, shortage of medication at the health centre/hospital, and a lack of transport) when trying to access medical help (**Figure 4C**).

MAPPING HUMAN–WILDLIFE INTERACTIONS AND RISK HOTSPOTS

Human, chimpanzee and environmental data are combined to identify areas of high spatial overlap between humans and chimpanzees, with increased likelihood of negative interactions (**Figure 5**). At the landscape level, risk areas constitute all areas where chimpanzees are recorded using space more intensively (**Figure 5A**) including villages and settlements within that territory. At the fine scale, risk sites constitute sites where people share resources with chimpanzees, including water, human crops and wild foods (**Figure 5B**).

DEVELOPING EVIDENCE-BASED OUTPUTS

Public Health Campaign and Clinical Training

COVID-19 and Ebola health campaigns are being conducted across Guinea-Bissau, including in Cantanhez NP. To avoid campaigning about another infectious disease, leprosy, and risking low engagement with local communities, we chose to

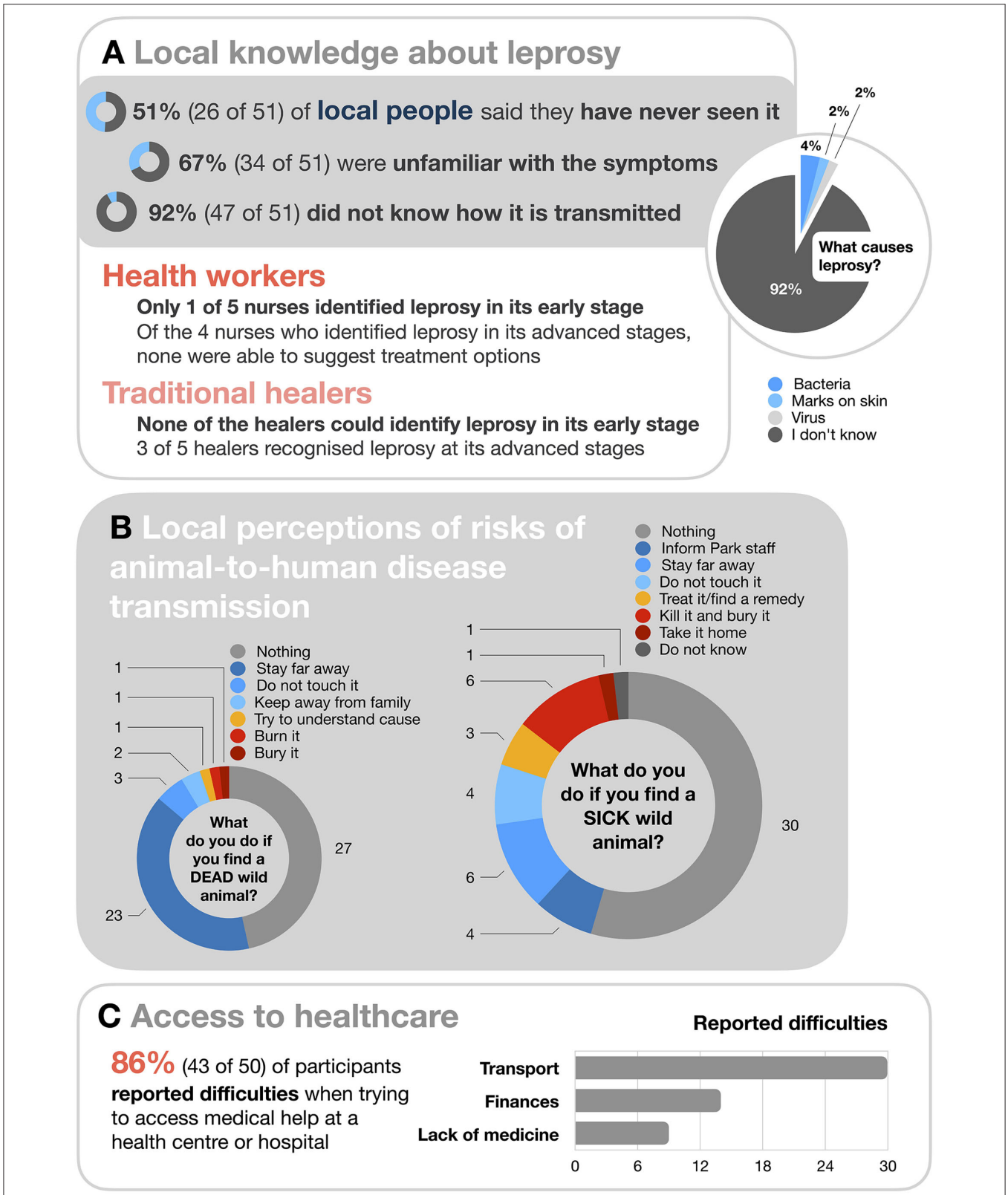


FIGURE 4 | Summary of results from interviews conducted across 27 villages in Cantanhez NP, to **(A)** understand local knowledge of disease including leprosy (51 household participants, five health workers and five traditional healers); **(B)** local perceptions of disease transmission risks (50 household participants); and **(C)** reported problems when attempting to access health care in Cantanhez NP (50 household participants). Values in sections **(B,C)**

(Continued)

FIGURE 4 | represent number of reported actions or difficulties, with some participants reporting more than one. Interviews were conducted in March and December 2020. We first piloted a semi-structured questionnaire with 31 participants across 13 villages, which included listing known infectious diseases and asking participants to describe symptoms, causes and treatment, as well as questions about disease in animals. To gather a knowledge baseline about leprosy, we subsequently conducted interviews across 27 villages, asking more specific questions about leprosy. For these we employed three semi-structured questionnaires: one for the general public ($n = 51$ participants), one for traditional healers ($n = 5$) and one for health workers ($n = 5$). To understand perceptions and mitigation of risks of infectious disease transmission and barriers when accessing healthcare, we employed a separate semi-structure questionnaire with 50 participants. We interviewed a total of 142 people, including 68 women and 74 men, from 10 ethnicities, including Nalu, Fula, Balanta, Tanda, Sussu, Djacanca, Pepel, Beafada, Bijagos, Mandinka.

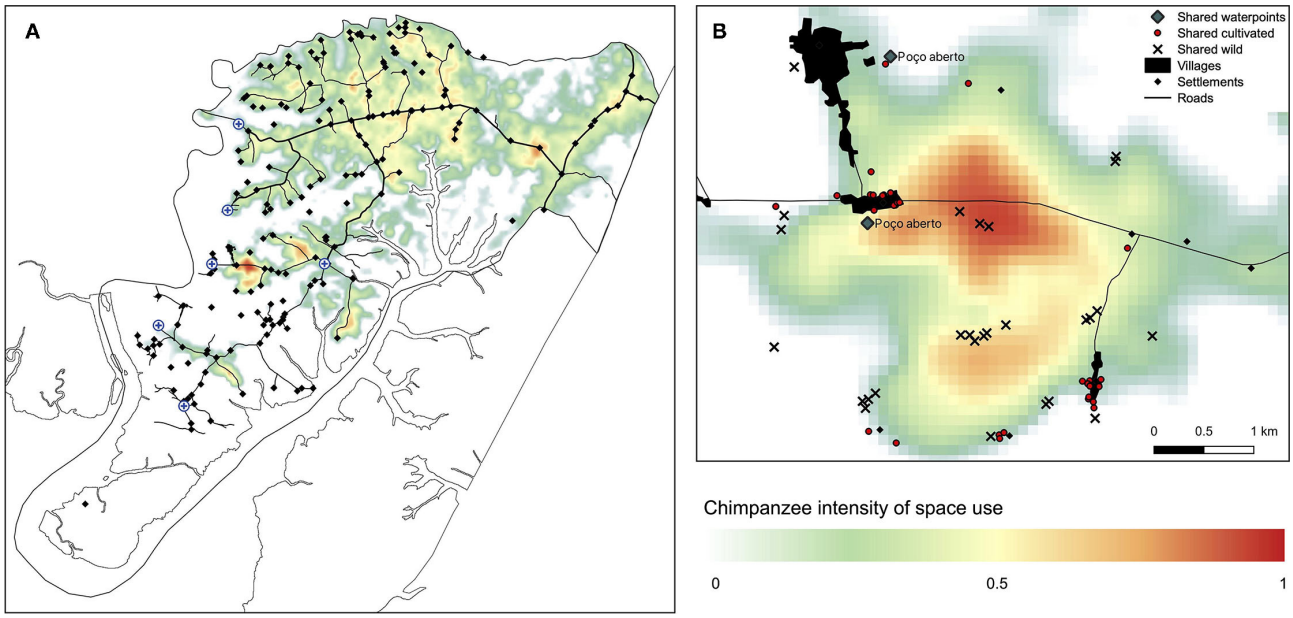


FIGURE 5 | (A) Preliminary risk map shows landscape-level intensity of space use by chimpanzees modelled using camera trap data and human features including roads, villages (black diamonds), and health centres (blue cross symbol); **(B)** an example of a fine-scale risk map showing use of space by chimpanzees (modelled using camera trap data) and humans (collected using participatory mapping) at Caiquene-Cadique. In June 2021, we began carrying out participatory mapping across 10 villages and 9 settlements in the central part of Cantanhez National Park using Cybertracker. The data model includes the following categories: wild and cultivated resources shared by people and wildlife, water points, direct encounters with wildlife by residents, activities associated with direct handling of wild animals (including hunting, processing animal products and keeping wild animals as pets), and reports of encounters with dead and "sick" animals.

develop a campaign to publicise One Health messages more broadly with leprosy used as one example of possible zoonotic infection. Delivering One Health concepts is complex; to reduce the risks of miscommunication regarding the disease threat posed by wildlife and the possible dangers to human health, we opted to reduce the geographical scale of the campaign to initially trial messages in selected villages. The focus on only six selected villages enables us to explore and identify effective ways to deliver One Health messages which will be incorporated into a One Health campaign strategy report for future application. Pre- and post-campaign interview questionnaires with participants from the selected villages will be used to evaluate changes in understanding of the links between environmental, animal and human health (see Measurable Indicator for Output 1, **Figure 2** caption). Additionally, working alongside Cumura hospital and AIFO, we will involve health workers as well as local health agents and healers in clinical training to identify all stages of leprosy disease with clear instructions for the referral protocol.

Biodiversity and Health Monitoring Programme

The BHMP has achieved increased wildlife monitoring capacity in Cantanhez NP through intensive training of local staff in data collection, including deploying and maintaining camera traps, recording direct and indirect animal observations and measuring habitat type using datasheets, GPS and Cybertracker. Guinean researchers are trained in data entry, management and analysis to ensure the sustainability of the monitoring programme. To promote good practise, staff have received training in the necessary health precautions during data collection.

One Health Environmental Management Plan

A One Health environmental management plan will promote healthy human–wildlife coexistence and strengthen multi-stakeholder decision-making capacity in Cantanhez NP. The plan aims to reduce possible risks of disease transmission within and

between-species and to inform future management strategies, including replanting and re-organising chimpanzee food sources across risk areas to reduce spatial overlap and conflicts. Key wildlife habitat and corridors, as well as areas of high human–wildlife interactions and potential disease transmissions are identified (see Measurable Indicator for Output 3, **Figure 2** caption) using data from the BHMP and participatory mapping. With additional GIS layers, including 2017–2021 vegetation cover and health landmarks (graded health centres, health units, healers), these data will help inform the development of the plan. Maps will be shown and discussed during meetings with local stakeholders, including the management committee, women's groups, farmers, and hunters. Maps will also be discussed with local people to identify aspects of human–wildlife interactions related to One Health that will need to be taken into consideration, such as sites where resources are shared (water points, cultivated foods). We use this opportunity to ask people to propose measures to minimise contact with wildlife. These meetings will be used to inform the content of the One Health environmental management plan.

Multi-Stakeholder Leprosy Response Plan

An effective infectious disease response plan requires coordination, communication and collaboration amongst conservation and health stakeholders. An increase in multi-stakeholder knowledge of leprosy occurrence in humans in Cantanhez NP and how to respond to conservation conflicts over disease will be achieved through multi-stakeholder collaboration in developing a leprosy response plan. The plan will be developed from published literature, knowledge exchange between project partners, including consideration of the socio-political context in Cantanhez NP, and the revision and development of a communication chain from local to national level (Cumura Hospital) and WHO. To reduce the risk of pathogen transmission and increase scientific knowledge on other infectious diseases present in Cantanhez NP, the response plan also involves the development and implementation of a protocol for the handling and management of dead chimpanzees and other wildlife. The plan will involve establishing a communication chain for when animal carcasses are found in the forest or information is received about a retaliatory wildlife killing. Information will need to be shared from residents to park guards, and then to IBAP managers and the Cantanhez Chimpanzee Project. Carcass swabbing, which involves taking biological samples from a carcass by inserting a sterile swab into natural body orifices or wounds, will then be carried out exclusively by trained staff using the appropriate PPE (FFP3 mask, goggles, and gloves). A detailed protocol for data and sample collection, decontamination, and waste disposal will be followed.

DISCUSSION

This project integrates biodiversity conservation and public health in response to the identification of leprosy in Critically Endangered chimpanzees and adopts a One Health approach to understand and tackle infectious diseases. Our causal pathway approach helped our team to identify our long-term One Health goals at Cantanhez NP and the conditions needed to achieve those goals, leading to better planning. Maintaining and strengthening the collaborative environment for project partners has been crucial for every aspect of this project. By investing in capacity and engagement of local communities and partners our goal is to strengthen the sustainability of the work. Our activities are designed to increase future resilience through training and capacity building so that in the potential absence of future funding, Bissau-Guinean partners and local collaborators have a strengthened knowledge base from which to build and continue key activities. From the outset we have ensured participatory community involvement in public health issues to reduce gender inequality, and strengthen synergy in stakeholder (local communities, Government, NGO) decision-making, and this will be an ongoing process.

Our project tackles many of the major challenges outlined in the WHO Global Leprosy Strategy 2021–2030, including cutting-edge research on wildlife and human dimensions of leprosy; reducing delays in detection; strengthening capacity and leprosy expertise; ensuring meaningful engagement of relevant stakeholders; reducing stigma in healthcare settings through knowledge and training; developing a communication and surveillance system; and providing suggestions for information systems to report leprosy cases. In particular, the strategy states that zoonotic transmission of leprosy until now appears to be low risk and highly localised. Our research provides up-to-date information on leprosy in human's closest living relative, the chimpanzee. Our ongoing work will provide key information on the potential for leprosy transmission and exchange between humans and wildlife, and between different wildlife species. Ultimately, successful, sustained, and effective disease control is likely to require more generalised poverty reduction and health capacity strengthening operations in affected nations. Applying a One Health approach, by tackling the interconnected threats to community and ecosystem health, has huge potential to advance responses to community and ecological health threats, mitigate conflict and limit the risk of infectious disease and its transmission.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the Ethics Committee of the University of Exeter Ref eCORN002528 v3.1. Verbal consent was obtained from interview participants, but written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements. The animal study was reviewed and approved by Ethics Committee of the University of Exeter Ref eCORN002520 v3.1.

AUTHOR CONTRIBUTIONS

All co-authors contributed to the development of the concepts presented in the paper and discussed and developed the Project Outputs. Data were collected by EB, MJ, AS, SC, SQ, and AL with support from KH, HP, AF-M, AR, ARS, QQ, SJ, LP, MR, JB, BG, CB, and FL. Data were analysed by EB, HP, AF-M, MJ, AS, and KH. EB and KH led on the writing of the manuscript, supported by all co-authors. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.735367/full#supplementary-material

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Drivers of Predator Killing by Rural Residents and Recommendations for Fostering Coexistence in Agricultural Landscapes

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Predators inhabiting human-dominated landscapes are vulnerable to various anthropogenic actions, including people killing them. We assess potential drivers of predator killing in an agricultural landscape in southern Chile, and discuss the implications for policies and interventions to promote coexistence. We evaluate five different types of motivation: (i) sociodemographics and household economy; (ii) livestock loss; (iii) predator encounter rates; (iv) knowledge of legal protection (all native predators are currently protected); and, (v) tolerance to livestock predation. As the killing of native predators is illegal, the prevalence of this behavior by rural residents was estimated using a symmetrical forced-response randomized response technique (RRT), a method designed to ask sensitive questions. A total of 233 rural residents from randomly assigned sample units (4 km^2) across the study region completed our questionnaire. More conspicuous species, such as hawks (*Falconiformes sp*), foxes (*Lycalopex sp*) and free-roaming domestic dogs (*Canis lupus familiaris*), were killed by a higher proportion of farmers than more cryptic species, like the felid güiña (*Leopardus guigna*), skunk (*Conepatus chinga*) and pumas (*Puma concolor*). The proportion of respondents admitting to killing predators was highest for hawks (mean = 0.46, SE = 0.08), foxes (mean = 0.29, SE = 0.08) and dogs (mean = 0.30, SE = 0.08) and lowest for güiña (mean = 0.10, SE = 0.09), which is the only species of conservation concern we examine (considered Vulnerable on the IUCN Red List). From our five motivation categories, past killing of predators was associated with higher reported predator encounter rates (guina, hawks), lower tolerance to livestock predation (hawks, dogs), higher reported livestock loss (dogs) and sociodemographics and household economy (foxes). Our results demonstrate that a one-size-fits-all approach to predator persecution is unlikely to reduce or eliminate illegal killings for the suite of species we examined. We identify and describe two main types of intervention that could foster coexistence, improvement of livestock management and domestic dog management in rural areas, as well as discussing the potential for social marketing.

Keywords: carnivores, free-roaming dogs, illegal behavior, *Leopardus guigna*, livestock predation, random response technique, tolerance to predation

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INTRODUCTION

Predators inhabiting human-dominated landscapes are vulnerable to a diverse range of anthropogenic activities, such as land-use change, habitat degradation, hunting for meat/trade and direct persecution in retaliation for livestock predation or due to cultural norms (Ceballos and Ehrlich, 2002; Treves and Karanth, 2003; Cardillo et al., 2004; Woodroffe et al., 2005; Inskip and Zimmermann, 2009). Carnivorous mammals and birds of prey are particularly susceptible to such impacts because of their slow life histories (Purvis et al., 2000). Historically, human persecution of predators has been responsible for species population declines and contributed to extinction events (Woodroffe, 2001). To enhance predator conservation, the motivations underpinning human induced mortality need to be identified and reduced so populations may persist and recover in the long-term (Treves and Karanth, 2003).

Human-predator coexistence can be achieved when the "interests of humans and wildlife are both satisfied" (Frank, 2016; Marchini et al., 2019). The outcomes of human-predator interactions are primarily determined by two main components: (i) how humans and predators interact; and, (ii) how humans react to those interactions (Marchini et al., 2019). At their worst, these interactions result in the killing of predators. Planning for coexistence therefore entails navigating away from such a response and toward more positive outcomes for predators and people (Frank, 2016). A first step in this process is understanding what drives a person to behave in a particular way, including why they kill predators (St John et al., 2013; Marchini et al., 2019).

How humans choose to act toward predators is influenced by factors operating at intertwined social (e.g., institutions, norms) and individual levels (e.g., sociodemographic and socioeconomic characteristics, attitudes, beliefs), (Lischka et al., 2018). It is thus important to understand individual-level factors that motivate the killing of predators (St John et al., 2015). In this study, we explore how five different types of motivation relate to predator-killing behavior by rural inhabitants in southern Chile. Our five categories, and the rationale for their inclusion, are as follows:

(i) *Sociodemographics and household economy:* There is evidence that behavior toward predators can be influenced by factors including education level, age and gender (Dickman et al., 2013). For example, poor rural inhabitants with few livelihood alternatives reported hostility toward predators (Naughton-Treves and Treves, 2005), and wealth was associated with higher tolerance for predators amongst commercial ranchers in Kenya (Romanach et al., 2007). Understanding how sociodemographics and household economy relate to predator-killing behavior can facilitate the targeted delivery of mitigation strategies through, for example, audience segmentation (Jones et al., 2019).

(ii) *Reported livestock loss:* The economic impact of livestock predation may cause the persecution of carnivores (Inskip and Zimmermann, 2009; Dickman, 2010; Marchini and Macdonald, 2012; Dickman et al., 2013). Where livestock losses are positively associated with predator persecution, interventions such as predator safe enclosures can be implemented to reduce predator impacts.

(iii) *Predator encounter rates:* The opportunity to kill a predator arises when the predator and human co-occur (Marchini and Macdonald, 2012; Carter et al., 2017). If reported predator encounter rates are positively related to predator killing behavior, and if these encounters occur near livestock enclosures, the provision of technical interventions (e.g., sound or lights to scare wildlife) could be a viable solution to reduce encounters. Also, as the outcome of this encounter depends also on how humans react, behavioral change strategies can be considered.

(iv) *Knowledge of legal protection:* Rules governing the acceptable use of, and interactions with, wildlife are central to natural resource management. Although the existence of rules alone does not guarantee compliance (Keane et al., 2008), knowledge of them can encourage it (e.g., Rizzolo, 2020, 2021). If this is the case, increasing awareness of regulations may reduce levels of illegal predator killing, particularly in areas where knowledge of the rules is limited.

(v) *Tolerance to livestock predation by specific predators:* People may express tolerance for wildlife by refraining from opposing conservation management and being willing to accept damage caused by wildlife (Frank, 2016). Here we use scenario-based questions to assess how farmers would respond to different levels of livestock loss by a range of predators. Furthermore, we investigate how this measure of tolerance relates to past predator killing behavior by the respondents. This sort of information can help target interventions toward the least tolerated species.

Planning for coexistence requires assessing human predator interactions at large spatial scales which is a significant challenge (Marchini et al., 2019; IUCN, 2020). Working across an extensive agricultural landscape, we estimate the proportion of rural residents who have killed nine legally protected predators, and compare these evaluations to two wild introduced species that people are permitted to control via lethal means. We also consider free-roaming domestic dogs (*Canis lupus familiaris*) because they are the main cause of livestock losses in Chile (Montecino-Latorre and San Martín, 2018). To address the challenge of asking questions about people's involvement in illegal acts, we use a symmetrical forced-response randomized response technique (RRT; Warner, 1965), a method designed explicitly for asking sensitive questions that has been used in a range of conservation settings (e.g., Razafimanahaka et al., 2012; St John et al., 2012; Gálvez et al., 2018). We examine how killing behavior can vary between species and implications this has for the design of interventions aimed at promoting sustained human-predator coexistence.

MATERIALS AND METHODS

Study Region and Sampling

Our study was conducted in the Araucanía region in southern Chile (**Figure 1**), just at the northern limit of the South American temperate forest ecoregion (39°15′S, 71°48′W) (Armesto et al.,

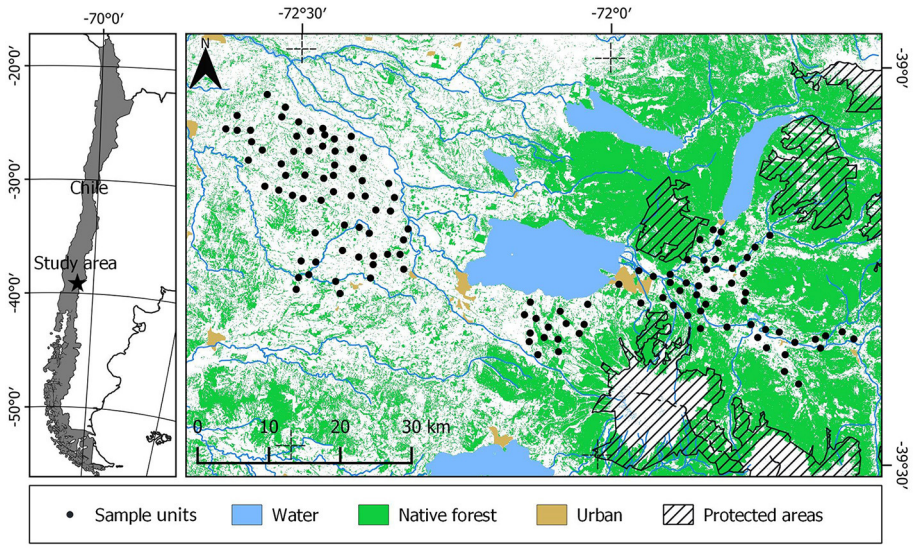


FIGURE 1 | Survey area in La Araucanía region of southern Chile. One or two households were surveyed within each of the 145 sample units (4 km^2) distributed across the study region. White background is mainly agricultural land use. Major water bodies correspond to Villarrica, Caburgua and Colico lakes. Protected areas are shown at high elevation sites within the Andes mountain range.

1998). The region comprises two distinct geographical sections common throughout southern Chile: the Andes mountain range and the central valley. Land-use in the latter is primarily intensive agriculture (e.g., cereals, livestock, fruit trees) and urban settlements. In the Andes, the agricultural lands become less intensively farmed (i.e., extensive livestock production and forestry) and are located within narrow valleys surrounded by continuous forest tracks on high slopes, which also include protected areas. The study region was divided up into a grid of 4 km^2 potential sampling units (SUs), representing a gradient of forest habitat fragmentation due to agricultural use and human settlement below 600 m.a.s.l. A total of 145 SUs were selected at random from the 230 in the grid, with 73 and 72 located in the central valley and Andes Mountains respectively.

Study Species

Our questionnaire focussed on predators that occur across the study region and that hunt small domesticated ruminants and/or poultry: (i) puma (*Puma concolor*), the largest predator present in Chile and known to predate ruminants (Murphy and Macdonald, 2010); (ii) güiña (*Leopardus guigna*), the smallest wild felid in the neotropics with a distribution restricted primarily to Chile and known to predate poultry (Sanderson et al., 2002; Gálvez et al., 2013); (iii) culpeo fox (*Lycalopex culpaeus*), a canid which will predate both small ruminants and poultry (Macdonald and Sillero-Zubiri, 2004); (iv) chilla fox (*Lycalopex griseus*), another canid which will predate both small ruminants and poultry (Macdonald and Sillero-Zubiri, 2004); (vi) Harris hawk (*Parabuteo unicinctus*); (vi) variable hawk (*Geranoaetus polyosoma*); and, (vii) Chilean hawk (*Accipiter bicolor*). All the raptors are known to predate poultry (Jimenez, 1986). To reduce the potential bias associated with respondents misidentifying species, we treated both canid species as "foxes" and all diurnal birds of prey as "hawks" in the analyses. Additionally, we included: (viii) the lesser grison (*Galictis cuja*), reported to predate on poultry (Silva-Rodríguez et al., 2007); and, (ix) Molina's hog-nosed skunk (*Conepatus chinga*), which is considered a nuisance, rather than a predator, of livestock. All nine species are protected by Chilean law (Agricultura, 1998), meaning that hunting them is prohibited. Only the güiña is of conservation concern and classified as Vulnerable by the IUCN Red List (Napolitano et al., 2015).

To examine and compare the killing behavior of respondents when a species is not legally protected, we also include all large mammals occurring in the study region that people are allowed to hunt without restriction. These comprise three introduced species: (i) hare (*Lepus capensis*); (ii) rabbit (*Oryctolagus cuniculus*); and, (iii) wild boar (*Sus scrofa*). Once again, we group the hare and rabbit together for analyses and refer to them as "lagomorphs." We also include free-roaming domestic dogs, which are an increasing problem in rural areas as they predate on wildlife (Sepúlveda et al., 2014) and livestock, especially small ruminants (Montecino-Latorre and San Martín, 2018). Currently, dogs are not mentioned specifically in the Chilean hunting law, which is the only legislative tool that classifies species as either protected or permitted to hunt (e.g., introduced or pest species).

Questionnaire Development and Delivery

The aim was to solicit information from rural inhabitants of the study region regarding their demographics and household economy, reported livestock loss, predator encounter rates, knowledge of legal protection of predators and tolerance to

predation by specific predators. The questionnaire consisted of six sections. The first of these comprised sociodemographic and socioeconomic questions relating to age (years), amount of schooling (years of school and college education), livelihood activities (categorical) and monthly household income (USD; continuous). Before the data were analyzed, the dependency of residents on agricultural activities undertaken on their land parcel for their livelihood was converted into one of three categories: 1 = no dependency; 2 = partial dependency (i.e., maintained some crops and domestic animals but also had income from another sources); and, 3 = complete dependency.

The second part of the questionnaire consisted of questions regarding killing predators. Because of the sensitive nature of the questions, we employed a symmetrical forced-response RRT design (St John et al., 2010, 2012; Ibbett et al., 2021), using a die as the randomization tool. Before answering each question, respondents rolled the die and were asked to provide a truthful answer if they rolled a one, two, three or four; answer "yes," irrespective of the truth, if they rolled a five; or, answer "no," irrespective of the truth, if they rolled a six. A physical barrier, consisting of a folder, was used to block the interviewer's view of the die, so the number could not be seen. Before moving onto the sensitive questions, trial RRT questions were conducted with respondents using non-sensitive questions to ensure they understood the instructions. After piloting, 10 years was deemed to be an appropriate recollection period.

The third questionnaire section asked respondents to report livestock losses to predation over the past year, or an alternative period they could quantify (e.g., per week, per month, per year), which we could later convert to an annual measure. If the alternative period was less than a year, the respondent was asked to give an average value (e.g., average losses per week for a year). In the fourth section, participants were probed about how frequently the species were encountered, once again allowing respondents to report a time period they could relate to. We then asked if they thought the hunting of each species was permitted or illegal according to the hunting law in Chile (Agricultura, 1998). Prior to analysis, their responses were coded as: 0 = thought hunting of the species was prohibited; 1 = did not know; 2 = believed hunting of the species was permitted with the expectation that perception of legality would increasingly influence the killing of predators. These responses were further coded as either correct or incorrect according to the hunting law, representing whether or not their knowledge of the law was accurate.

The fifth section of the questionnaire presented farmers with scenarios to evaluate their tolerance to predation caused by different specific predators. Respondents had to state how they would respond to partial predation of a livestock holding of either 100 sheep or 100 chickens, depending on the predator. Respondents were asked what behavior they would display toward the predator after the loss of 2, 10, 25, 50 and >50 sheep or chickens had been experienced. For sheep predation, we assessed puma and domestic dogs, and for chicken predation we asked about güiña and "hawks." Response options included: (a) doing nothing; (b) improvement of livestock management through the use of enclosures; (c) calling wildlife authorities to alert them to the presence of the predator; (d) non-lethal capture of predator and handover to the authorities; (e) use of predator deterrents; and, (f) control via killing directly (i.e., the householder would kill the predator rather than requesting assistance from the authorities). Prior to analysis, we grouped the scenario responses into three categories of increasingly negative behavior toward predators: 0 = would remain passive and do nothing (item a); 1 = would carry out some sort of non-lethal or active management (items b-e); and, 2 = would carry out lethal control of the predator (item f). To assess if householders had access to the necessary skills and equipment required to hunt predators, we asked participants whether anyone in the household participates in sport hunting (a legal activity in Chile, which includes the use of snares, and can be conducted with a license that is inexpensive to obtain).

The final part of the questionnaire asked current management of livestock, particularly sheep and chickens. For example, we asked if the household enclosed livestock at night, the distance of the enclosure from the house, the number of domestic dogs/cats associated with the property and how they are managed overnight (e.g., free-roaming, tethered), as well as how often they are fed (meals per unit time) and the type of food they are given (categorical).

The questionnaire was piloted with 10 households occurring outside of the SUs, with one individual completing it on behalf of the entire household. The feedback from the pilot was used to improve the wording (e.g., the hypothetical question was refined to maximize clarity), time-scale (e.g., 10 year recall period) and order of questions (e.g., to make the flow of the questionnaire as logical as possible for the respondents). The data collected from the pilot were discarded.

The final questionnaire (**Appendix S1** in **Supplementary Material**) was administered face-to-face with a household representative, with one or two households sampled per SU, during May to September 2013. Questionnaires were administered by NG who is Chilean and has lived in the study region for over 10 years. The gender of the household representative was dependent on the individuals present when the household was approached and who appointed themselves the representative. Due to the traditional roles of males in rural Latin-American societies in relation to dealing with outsiders and/or officials, our sample was predominantly male (80%). Overall, the sampling strategy covered 66% of the households within each 4 km^2 SU in the study region (Gálvez et al., 2018).

The study was approved by the School of Anthropology and Conservation Research and Research Ethics Committee at University of Kent, and the Pontificia Universidad Católica Ethics committee. Data collection was anonymous and free prior informed consent was sought from all participation.

Data Analysis

Data analyses were conducted in R version 3.2.0 (R Core Team, 2015). The proportion of respondents admitting to killing each predator species was estimated using the model of Hox and Lensvelt-Mulders (2004):

$$\pi = \frac{\lambda - \theta}{s}$$

TABLE 1 | Sociodemographic and socioeconomic characteristics of questionnaire respondents living within the Araucanía region of southern Chile ($N = 233$).

Sociodemographic and socioeconomic characteristics	Mean	SE	Median	Minimum	Maximum
Property size (ha)	98	0.85	29	1	1,200
Time living at the property (years)	35	0.09	35	1	87
Age (years)	56	0.06	55	22	87
Amount of schooling (years)	10	0.01	10	0	18
Household income (USD per month)	558	2.81	341	59	5,934
No. of small ruminants	14	0.07	10	0	170
No. of chickens	23	0.09	18	0	120

where π is the estimated proportion of people in the sample who have undertaken the behavior, λ is the proportion of respondents who said "yes," θ is the probability of the answer being a forced "yes," s is the probability a respondent had to answer the question truthfully. A total of 10,000 bootstraps samples were run to calculate 95% confidence intervals, accounting for sample and RRT method uncertainty. All continuous predictors were z-transformed to standardize the scale of effects. Before exploring which of our explanatory variables may predict killing behavior for each specific predator, we checked them for collinearity using a Spearman's rank correlation coefficient matrix. Variables with Spearman's $|\text{rho}| > 0.7$ were removed from the analysis. Between correlated variables, we left those that were easier to interpret. For example, years of schooling, farm size and hunting for sport were correlated with income, thus leaving the latter as an easier value to interpret. Similarly, the use of night enclosures for chickens and sheep were correlated with land parcel dependency for their livelihood. Consequently, a total of eight potential predictors were retained from our five categories: (i) age, annual household income, the dependency of residents on livestock holdings (sheep or chickens) and crops on their land parcel for their livelihood; (ii) reported livestock loss (reported animals lost/year); (iii) reported predator encounter rates; (iv) knowledge of legal protection; and (v) tolerance to livestock predation by specific predator.

The RRlog function in the R package RRreg (Heck and Moshagen, 2018) was used to conduct multivariate logistic regression using the model for a symmetrical forced-response RRT data. For each predator, we fitted a logistic regression model with the potential predictors of killing behavior and evaluated their significance with likelihood ratio tests (LRT ΔG^2). First, a full model (i.e., all eight predictors) was evaluated. Full models of güiña and domestic dogs had convergence problems or generated nonsensical estimates (e.g., p-value of 1). We removed variables from the full model, in a backward manner, to identify predictors that triggered extreme estimates. Simultaneously, we conducted a univariate analysis of each predictor, as well as a multivariate subset of predictors to evaluate stability of estimates and consistency regarding significance and direction of relationships. We retained predictors in the model that allowed stability. None of the excluded predictors resulted in significant estimates in either the univariate or multivariate subset models. Only sociodemographic and household economy predictors were excluded across all the predator models and this was because their inclusion created instability (**Appendix S2** in **Supplementary Material**).

RESULTS

The questionnaire was completed in full by 233 rural residents (response rate of 99% of households approached) living within the study region (**Table 1**). Most respondents were male (80%), had grown up in a rural area (80%) and lived at their property full-time (97%). One farm was very large (1,200 ha), but most were considerably smaller (median = 29 ha). Respondents had 10 years of formal schooling on average, with 50% having received between 7 and 12 years of education. A high percentage of respondents (82%) reported that their dogs were left free to roam at night and the mean number of dogs per household was 3 (SE = 0.01; range = 1–28).

Pumas, güiñas and the lesser grisson were rarely encountered by respondents, while hawks and lagomorphs were frequently observed. Indeed, most of the rural residents reported seeing lagomorphs and hawks everyday (**Table 2**). Most respondents knew how the hunting law related to each species, with the exception of free-roaming domestic dogs that were perceived incorrectly as being protected by the hunting law (**Table 2**).

The reported predators of sheep were puma (43% of respondents had experienced livestock loss via this species), domestic dogs (41%) and, to a much lesser extent, foxes (6%). The number of sheep killed per year was similar across predators, with most respondents stating <10 are lost on average (**Figure 2A**). However, there were some outliers where dogs had killed substantial numbers of sheep. The main reported poultry predators were hawks (75%), foxes (50%) and güiña (16%), with the reported number of animals predated per year highest for hawks and foxes (**Figure 2B**).

Across all the scenarios designed to measure tolerance to the predation of livestock holdings, a significantly larger proportion of respondents said they would kill free-roaming domestic dogs, compared to pumas (**Figure 3**). Moreover, compared to other predators, the proportion of rural residents stating that they would kill domestic dogs was relatively high (>0.6). For all species, the rate of increase in the proportion of respondents

TABLE 2 | Questionnaire respondents' ($N = 233$) knowledge of how the hunting law in Chile relates to each of the predators in our study, and the frequency of encounters they have with each species on their property or surroundings.

Species	IUCN red list status	Hunting is legally permitted	Respondents' knowledge of legal hunting status for each species (%)		Respondents' reported encounters with species (per year)	
			Correct	Do not know	Mean (SE)	Median
Puma	LC	No	99	1	1.8 (0.02)	0.2
Güiña	V	No	79	17	0.2 (0.00)	0.0
Foxes	LC	No	94	3	41.2 (0.34)	12.0
Hawks	LC	No	78	15	204.0 (0.70)	360.0
Molina's hog-nosed skunk	LC	No	70	20	23.7 (0.21)	12.0
Lesser grison	LC	No	62	30	2.8 (0.10)	0.0
Domestic dog	–	Not included in hunting law	28	26	81.8 (0.57)	12.0
Lagomorphs	–	Yes	77	10	319.0 (0.45)	360.0
Wild boar	–	Yes	55	13	6.4 (0.11)	0.0

The International Union for Conservation of Nature (IUCN) Red List status is provided for each predator as an indication of conservation status. "Foxes" refers to both culpeo (Lycalopex culpaeus) and chilla foxes (Lycalopex griseus). "Hawks" refers to all diurnal birds of prey. "Lagomorphs" refers to rabbits and hares.

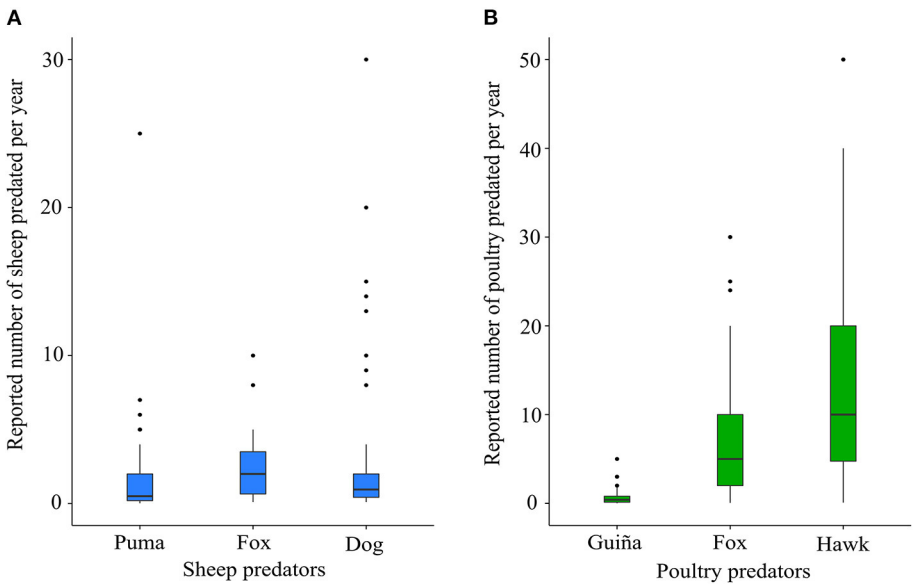


FIGURE 2 | The number of reported **(A)** sheep and **(B)** poultry killed by predators per year provided by rural inhabitants of the Araucanía Region of southern Chile ($N = 233$).

stating that they would kill a predator was greatest between 2 and 25 livestock killed, remaining constant for >25.

The proportion of respondents who reported killing predators via RRT varied across species (**Figure 4**). For puma, the 95% confidence intervals overlap zero, suggesting that the behavior may not have occurred in the past decade, or that the occurrence was very low. Only a small proportion of rural residents (10%) report killing güiña, while estimates for domestic dogs, foxes and hawks were greater (30–40%). There were large differences in the proportion of respondents reporting hunting legally; many hunted lagomorphs whilst few hunted wild boar.

Factors associated with killing behavior reported via RRT varied by species (**Table 3**). The probability that a respondent had killed güiña or hawks increased with encounter frequency (güiña $\beta = 0.86$, se $= 0.63$ $p = 0.04$; hawk $\beta = 0.62$ se $= 0.30$, $p = 0.04$), whereas the likelihood of fox killing rose with the extent of economic dependency the rural resident had on their land parcel ($\beta = 0.72$, se $= 0.35$, $p = 0.03$). Respondents who were less tolerant to predation were significantly more likely to report killing hawks in the case of chickens ($\beta = 1.07$, se $= 0.41$, $p = 0.004$) and dogs in the case of sheep ($\beta = 2.79$, se $= 1.88$, $p = 0.0003$). Reported loss of sheep was also positively and significantly related to reported dog killing ($\beta = 3.52$, se $= 1.74$, $p = 0.01$). The RRT data on puma killing were not modeled due to the exceptionally low prevalence of this behavior (**Figure 4**).

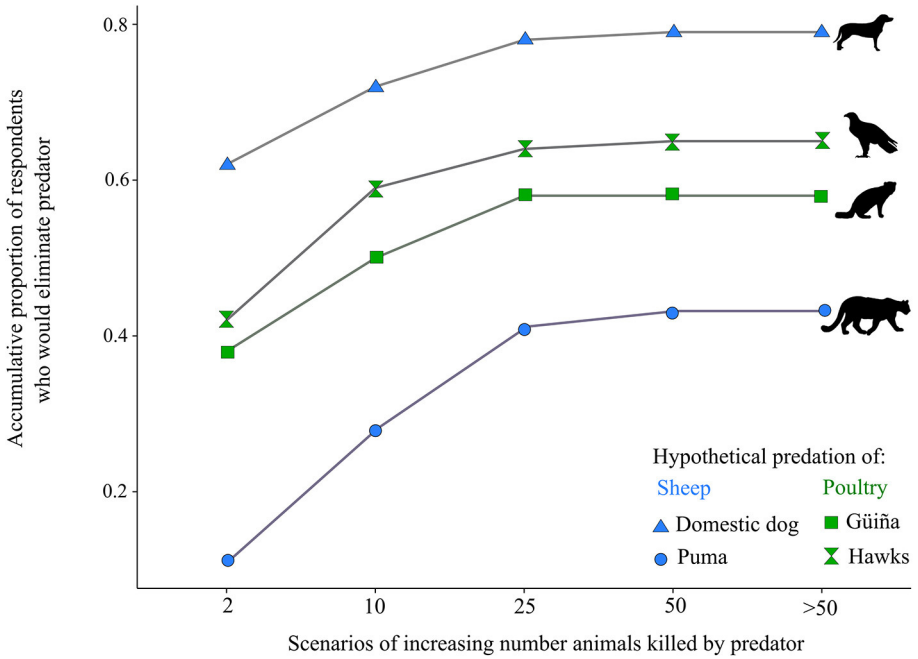


FIGURE 3 | Tolerance to livestock predation as the proportion of questionnaire respondents ($N = 233$) reporting that they would kill a specific predator in response to an increasing quantity of hypothetical livestock killed. The baseline for each scenario was that a farmer had a total of 100 sheep or chickens and experienced losses of 2, 10, 25, 50 and >50 individuals as a consequence of predation. The puma and domestic dog are sheep predators, whereas hawks and güiña are the poultry predators.

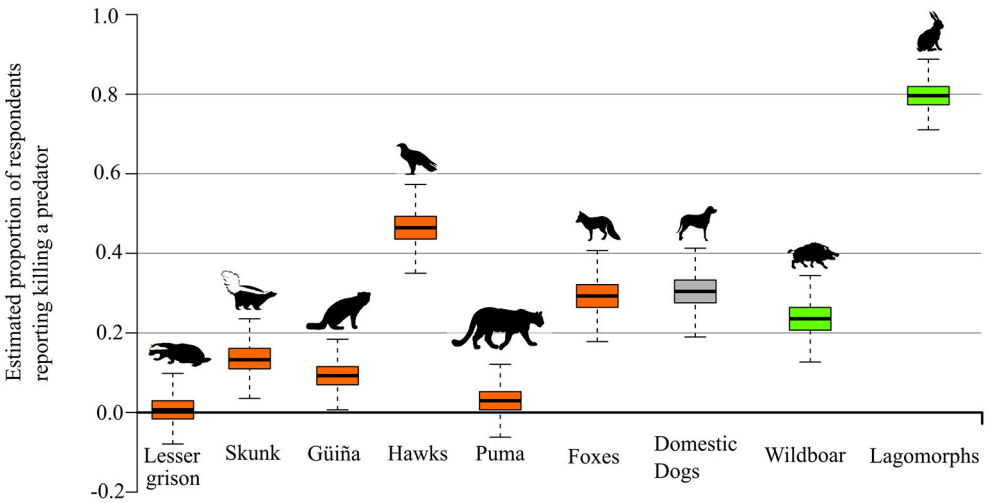


FIGURE 4 | The proportion of questionnaire respondents ($N = 233$) admitting to killing a species in the past decade. Values were estimated by the randomized response technique (RRT) and 95% confidence intervals were obtained from 10,000 bootstrapped samples. The species are grouped according to whether or not hunting is permitted under Chilean hunting law or not (illegal to hunt native species, orange; legal to hunt introduced species, green). Domestic dogs (gray) are not listed as either legal or illegal to hunt in the law.

DISCUSSION

Securing the long-term persistence of predator populations in human-dominated landscapes requires effective conservation management policies and interventions informed by evidence (Linnell et al., 2001). Our results highlight that a one-size-fits-all approach to minimizing persecution is unlikely to reduce or eliminate illegal killings across all the key predators in our study region. A high proportion of our respondents reported engaging in legal hunting (e.g., shooting of lagomorphs and

TABLE 3 | The relationship between illegal killing of predators and potential predictors of the behavior amongst questionnaire respondents ($N = 233$).

Species	Predictors	Coefficient	SE	*p*	Odds ratio	Odds ratio	
						Lower CI	Upper CI
Güiña	(Intercept)	−3.5	2.47	0.17	0.03	0	3.84
	Age[a]	–	–	–	–	–	–
	Income	0.23	0.33	0.52	1.26	0.66	2.4
	Land parcel dependency[a]	–	–	–	–	–	–
	Livestock holdings (chickens)	−1.24	1.48	0.35	2.95	0.45	19.19
	Knowledge of legal protection	1.08	0.96	0.53	0.29	0.02	5.28
	Reported encounter rate	**0.86**	**0.63**	**0.04**	**2.37**	**0.7**	**8.11**
	Reported livestock loss	0.1	0.43	0.81	1.11	0.48	2.56
	Tolerance to livestock predation	0.02	1.38	0.99	1.02	0.07	15.34
Hawk	(Intercept)	−2.04	0.96	0.02	0.13	0.02	0.86
	Age	0.42	0.31	0.13	1.52	0.83	2.78
	Income	−0.01	0.28	0.97	0.99	0.57	1.71
	Land parcel dependency	0.43	0.38	0.24	1.54	0.73	3.23
	Livestock holdings (chicken)	0.08	0.7	0.91	1.08	0.28	4.23
	Knowledge of legal protection	−0.02	0.53	0.97	0.98	0.35	2.77
	Reported encounter rate	**0.62**	**0.3**	**0.04**	**1.85**	**1.03**	**3.34**
	Reported livestock loss (chickens)	0.43	0.35	0.2	1.53	0.78	3.02
	Tolerance to livestock predation	**1.07**	**0.41**	**0.004**	**2.92**	**1.32**	**6.49**
Fox	(Intercept)	−2.30	0.89	0.00	0.10	0.02	0.58
	Age	−0.11	0.26	0.66	0.89	0.54	1.48
	Income	−0.38	0.53	0.33	0.69	0.24	1.93
	Land parcel dependency	**0.72**	**0.35**	**0.03**	**2.05**	**1.04**	**4.07**
	Livestock holdings (chicken)	0.10	0.24	0.66	1.11	0.69	1.79
	Knowledge of legal protection	−0.77	1.14	0.42	0.46	0.05	4.31
	Reported encounter rate	0.22	0.24	0.35	1.25	0.78	1.99
	Reported livestock loss (chickens)	−0.03	0.23	0.89	0.97	0.61	1.53
Dog	(Intercept)	−5.73	3.31	0.00	0.00	0.00	2.12
	Age	−0.55	0.37	0.10	0.58	0.28	1.19
	Income	−0.20	0.32	0.51	0.82	0.44	1.54
	Land parcel dependency	0.17	0.41	0.68	1.18	0.53	2.63
	Livestock holdings (sheep)[a]	–	–	–	–	–	–
	Knowledge of legal protection	0.28	0.34	0.41	1.32	0.68	2.60
	Reported encounter rate	0.06	0.38	0.87	1.06	0.50	2.24
	Reported livestock loss (sheep)	**3.52**	**1.74**	**0.01**	**33.62**	**1.12**	**1008.17**
	Tolerance to livestock predation	**2.79**	**1.88**	**0.0003**	**16.35**	**0.41**	**657.10**

[a] *These predictor variables were excluded from the full model (i.e., all predictors included) due to model instability (i.e., convergence or non-sensical estimates). None of the excluded predictors resulted in significant estimates in either the univariate or multivariate subset models. Reported coefficients, standard errors, odds ratios and their 95% confidence intervals were derived from a multivariate logistic regression which incorporates the known probabilities of the forced-responses obtained with the randomized response technique (RRT). We tested eight non-correlated predictors of five categories of predator killing behavior: (i) sociodemographic and household economy predictors—age (years), annual household income (USD), the dependency of the rural residents on their land parcel for their livelihood (1 = no dependency; 2 = partial dependency; 3 = complete dependency) and livestock holdings (sheep or chickens); (ii) reported livestock loss (animals lost/year); (iii) reported predator encounter rates (frequency of encounters/year); (iv) knowledge of legal protection (0 = hunting prohibited; 1 = do not know; 2 = hunting permitted); and (v) tolerance to livestock predation by a specific predator (0 = do nothing; 1 = manage predator; 2 = kill predator). All continuous variables were standardized to z-scores. Significance was at the p = 0.05 level and is indicated in bold. Hypothetical predation scenario was not included as a variable for foxes as the species was not included in this section of the questionnaire.*

wild boar), indicating that they are likely to possess the skills and resources to potentially kill predators illegally. We found that more conspicuous species, such as hawks, foxes and free-roaming domestic dogs, were killed by a higher proportion of farmers than more cryptic species, like güiñas, skunks and pumas. Indeed, from our five motivation categories, past killing of predators (i.e., yes/no) was associated with higher predator encounter rates for güiña and hawks, lower tolerance to livestock predation (hawks, dogs), higher livestock loss (dogs) and higher dependence of households on their land parcel (foxes). These drivers have implications for planning future coexistence interventions which we group into two main types: (i) improving livestock management; and (ii) better domestic dog management within rural areas.

The significant relationship between reported encounter rate and both güiña and hawk killing reported via RRT highlights the need for enhanced poultry management. The güiña is the only threatened predator that is found within the agricultural landscape, and it is probable that their low encounter rate explains the relatively low prevalence of killing. When presented with the hypothetical scenario of a güiña predating their chickens, many of the respondents reported that they would kill the offending animal. However, tolerance to livestock loss was not a significant predictor of respondents' past güiña killing behavior. This reflects the negative opinions rural residents have of güiña (Herrmann et al., 2013). Our results suggest that the prevalence of güiña killing would be higher if encounter rates were greater. People normally kill güiñas when they are caught inside the chicken coop (Sanderson et al., 2002; Gálvez et al., 2013) and hawks are usually killed when detected surrounding chicken enclosures. Managing poultry by housing them at night would minimize losses by the predominantly nocturnal güiña (Hernandez et al., 2015), while properly trained and managed guard dogs and the addition of tree cover around chicken enclosures could reduce hawk predation (Almuna et al., 2020).

Reported livestock predation levels only predicted respondents' predator killing behavior for free-roaming domestic dogs. Tolerance for livestock loss due to dogs was also significantly related to their dog-killing behavior as reported via RRT, and rural residents reported lower levels of tolerance for livestock predation by dogs than for all the other predators. For example, 62% of respondents reported that they would kill a dog if they lost just two sheep, whereas just 11% said the same for puma. Our findings, combined with anecdotal evidence from informal conversations with respondents, suggest that domestic dogs in rural areas are viewed negatively with respect to sheep predation, as is the case elsewhere in Chile (Villatoro et al., 2019). The extent of the issue was illustrated by the fact that dogs were reported to have predated sheep on more than 40% of the farms. Rural residents continually mentioned domestic dogs as their main livestock predation "problem," together with the perception that it was illegal to kill dogs according to the Chilean hunting law. At the time of data collection, many respondents (30%) reported killing dogs and the legal status of domestic dogs lacked clarity. However, in 2017, a new law came into force that was strongly lobbied for by animal rights groups which prohibited euthanasia and lethal control of domestic dogs irrespective of their involvement in sheep predation. The law should be revised as a way to dissuade the "shoot, shovel and shut up" dynamic which is likely to be occurring. The current situation aligns the desires of conservationists who are concerned that dogs kill wildlife in Chile (Silva-Rodríguez and Sieving, 2012) with that of rural residents, but increases tensions with animal rights groups. This emerging conflict requires resolution strategies (Redpath et al., 2013) that could improve dog welfare, reduce free-roaming behavior and, in turn, lead to a decline in livestock and wildlife predation. While sterilization programs may reduce dog births and population sizes over time, the main challenge in rural areas is to influence the social acceptability of free-roaming behavior and overcome peoples reticence to tether or restrict their movement (Villatoro et al., 2019 and shown in this study).

Over three quarters of our respondents knew how the hunting law in Chile related to each of the native predators. Nonetheless, across all species, respondents' knowledge of the law was not significantly related to their reported killing behavior. This suggests that, as observed in other studies (e.g., Rowcliffe et al., 2004), knowledge of laws neither guarantees compliance nor translates into tolerance for predators. The limited level of on-the-ground enforcement and thus low perceived risk of sanctions (Rowcliffe et al., 2004) may explain why some predator persecution still occurs in the study system. However, increased enforcement seems highly unlikely at present given budget restrictions for wildlife programs within the Ministry of Agriculture in Chile (Maldonado, 2018), making this an inefficient tool to reduce the killing of predators. Social marketing campaigns offer an alternative approach to encouraging behavior change. Well-designed, targeted and evaluated social marketing interventions can promote tolerance and coexistence (Veríssimo et al., 2019). For example, campaigns encouraging farmers to adopt predator deterrents could successfully reduce encounter probabilities (e.g., Ohrens et al., 2019; Almuna et al., 2020). Rather than purely disseminating information about prohibitive laws, messages that focus on what to do in case of encounters and the benefits associated with predator presence in landscapes (e.g., pest control) may improve tolerance (Slagle et al., 2013; Bruskotter and Wilson, 2014). The link between knowledge of benefits, tolerance and killing-behavior remains relatively understudied, but would be an interesting and potentially fruitful avenue for future research in this study system.

While randomized response techniques are reportedly harder for respondents to understand compared to other specialized questioning techniques (Davis et al., 2019), we deployed a symmetrical forced-response RRT design reputed for design efficiency (Lensvelt-Mulders et al., 2005) and encouraging more honest reporting of sensitive information (Ostapczuk et al., 2009). In recognition of the challenges associated with investigating sensitive topics, specialized questioning techniques such as RRT are increasingly being used in conservation to provide greater anonymity to respondents, improve response rates and decrease biases. However, a recent review of RRT applications in conservation provides evidence that whilst RRTs typically outperform direct questions in other disciplines, they do not yet do so in conservation (Ibbett et al., 2021). Prior to committing to incorporating any form of specialized questioning technique into a study, we encourage researchers to consider factors such as topic sensitivity, suspected prevalence (e.g., common or rare), achievable sample size and the type of estimate required (Nuno and St. John, 2015; Hinsley et al., 2019; Ibbett et al., 2021). Additionally, while our survey was conducted on behalf of the entire household, the majority of respondents were male. Our analyses and recommendations (e.g., targeted social marketing) may therefore not fully capture the role women play in predator persecution, as gender has been found to be important for understanding human-wildlife dynamics and conservation in other contexts (e.g., Agu and Gore, 2020). Nonetheless, in-depth qualitative investigations in our study region have shown that women can hold negative

perceptions toward wild predators when they predate on livestock (Benavides, 2020).

Our intensive data collection over a relatively large area provides important information at a scale necessary for planning and delivering coexistence. Identified drivers may also be used as surrogates or proxies for the actual killing of predators in some contexts. For instance, the hypothetical predation scenarios (i.e., tolerance to predation by specific predator; **Table 3**) for hawks and domestic dogs were related to reported killing behavior, suggesting that the scenario-based questions can be a useful proxy measure of involvement in sensitive acts. Meaningful engagement with people bearing the economic, physical, and psychological costs of predator-coexistence is crucial to navigate toward coexistence (Redpath et al., 2015; Pooley et al., 2016) and, while it may be impossible to eradicate the illegal killing of predators, increased tolerance becomes more viable once drivers of persecution are identified and tailored interventions are implemented. Identification of relevant drivers in this particular landscape offers conservationists a more targeted species-specific toolbox to inform the development of interventions, such as the importance of improved chicken enclosures for güiña and hawks, use of deterrents to reduce predator encounters, social marketing to improve outcomes when encounters do occur and to increase the social acceptability of restricting free-roaming dogs. Once implemented, the performance of the interventions need to be evaluated. Our work provides a baseline to assist in monitoring the prevalence of predator killing behavior.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the School of Anthropology and Conservation Research and Research Ethics Committee at University of Kent, and the Pontificia Universidad Católica Ethics Committee. Data collection was anonymous and free prior informed consent was sought from all participation. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

NG carried out field work, analyses, and elaborated figures. All authors designed the study and questionnaire instrument, and contributed significantly to the writing of the manuscript.

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SUPPLEMENTARY MATERIAL

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METHODS
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Planning for Human-Wildlife Coexistence: Conceptual Framework, Workshop Process, and a Model for Transdisciplinary Collaboration

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Coexistence, as a concept and as a management goal and practice, has attracted increasing attention from researchers, managers and decision-makers dedicated to understanding and improving human-wildlife interactions. Although it still lacks a universally agreed definition, coexistence has increasingly been associated with a broad spectrum of human-wildlife interactions, including positive interactions, transcending a conservation focus on endangered wildlife, and involving explicitly considerations of power, equity and justice. In a growingly complex and interconnected human-dominated world, the key to turning human-wildlife interactions into large-scale coexistence is thorough planning. We present an approach for evidence-based, structured, and participatory decision-making in planning for human-wildlife coexistence. More specifically, we propose (i) a conceptual framework for describing the situation and setting the goals, (ii) a process for examining the causes of the situation and creating a theory of change, and (iii) a model for transdisciplinary research and collaboration integrating researchers, decision-makers and residents along with the interests of wildlife. To illustrate the approach, we report on the workshop considering the *Jaguars of Iguaçu*, a conservation project whose strategy includes the improvement of the relationship between ranchers and jaguars outside Iguaçu National Park, Brazil.

Keywords: collaboration, conservation planning, human-wildlife conflict, stakeholder processes, systems thinking, theory of change, transdisciplinarity

INTRODUCTION

The management of human-wildlife interactions (HWIs), the effects of which include collisions with vehicles, damage to property and agricultural production, zoonotic diseases, and the use of animals as a resource, is becoming more challenging (Broad et al., 2014; Aguirre, 2017; Pooley et al., 2017; IUCN, 2020). Behind this trend are the rapid and profound changes in the physical environment and societal values associated with the Anthropocene and modernization, including factors such as climate change, expanding infrastructure, urbanization, economic globalization, the digital revolution, and the expanding scope of ethical considerations (Vucetich et al., 2021a). One view, that we share, is that in an increasingly complex and interconnected human-dominated world, turning HWI into large-scale coexistence requires thorough planning (Marchini et al., 2019). Nonetheless, goal-setting and decision making in HWI management has been slow to rise to this challenge, perhaps impeded by, amongst other things, the lack of (i) a conceptual framework that integrates wildlife and human goals in order to articulate precisely the desired changes, (ii) a structured, interdisciplinary approach to assess the situation, select actions, and measure their success so as to inform how to cause the changes effectively, and (iii) the proper integration of stakeholders–scientists, decision-makers and residents–to jointly choose the right changes and the means to promote them, and to work together to implement them.

Planning is the process which, when successful, identifies a course of action in a systematic manner to achieve objectives by utilizing the available resources competently in a cost-effective way (Mintzberg and Quinn, 1996; IUCN, 2017). The process starts by addressing the two fundamental questions: where are we and where do we want to get (i.e., what are the current and desired situations, respectively)? The fast-growing scientific literature on HWI provides a clear answer to these questions regarding the general direction to be taken: the aspiration is to transform "human-wildlife conflict" (IUCN, 2020) into "human-wildlife coexistence" (Frank, 2016; Frank et al., 2019). The challenge with this, however, is that "coexistence" is a very vague vision, and can mean many different things to different people in different contexts. The conceptualization and operationalization of solutions to human-wildlife coexistence is still a matter of debate (Carter and Linnell, 2016; Koenig et al., 2020; Glikman et al., 2021; Pooley et al., 2021).

Indeed, "coexistence" is a broad concept and usually too vague to provide a clear functional goal for a conservation initiative. Rather, it needs to be broken down into clear, specific, and achievable envisaged outcome appropriate for the given situation. With a clear objective in hand, the next guiding question to be addressed in the planning process is how to get there? A roadmap to human-wildlife coexistence has to be produced to guide the actions. Nonetheless, despite the wealth of knowledge about HWI generated in the last couple of decades (Nyhus, 2016; Frank et al., 2019; Koenig et al., 2020), and the diversification and dissemination of techniques and tools to enhance decision-making (Schwartz et al., 2018), many projects and programs dedicated to preventing and mitigating human-wildlife conflict (HWC) and/or promoting human-wildlife coexistence still:

(1) lack a clear theory of change informing the linkage between actions and expected effects,
(2) base decisions on unverified, and sometimes flawed, assumptions about those linkages, and
(3) evaluate success based on outputs directly produced by the actions (e.g., number of community workshops conducted) without the proper attention to the indirect, long-term effects (e.g., behavior change among workshop participants).

Evidence-based and structured decision-making in HWC and coexistence requires integration between researchers and decision makers. However, we argue that research in academia has had a strong emphasis on describing and explaining problems instead of testing solutions and measuring the associated direct and indirect changes (but see Van Eeden et al., 2018a,b; Sutherland et al., 2021). Projects and programs, in turn, have not used the scientific evidence available to guide actions and evaluate results to the extent they could. This gap between research and implementation has hindered effective and sustainable solutions (Knight et al., 2008; Toomey et al., 2017; Ferraz et al., 2020). Insufficient engagement of various stakeholders, such as the local residents, can also undermine efforts to improve HWI.

In this paper we present an approach for evidence-based, structured, and participatory decision-making in planning for human-wildlife coexistence. More specifically, we propose:

(i) a conceptual framework for describing the current situation of both wildlife and people in the context of their interaction, and the desired changes i.e. setting the goals,
(ii) a process for examining the causes of such situation and creating a theory of change (ToC), and
(iii) a model for transdisciplinary research and collaboration integrating researchers, decision-makers and residents.

To illustrate the approach, we report on a workshop conducted with Jaguars of Iguaçu, a project the goal of which is the conservation of the jaguar (*Panthera onca*) as a key species for the maintenance of biodiversity inside and outside Iguaçu National Park (Parque Nacional do Iguaçu, PNI), Brazil.

WORKSHOP PROCESS DEVELOPMENT AND CASE STUDY

The workshop process outlined here has been developed by the authors of this paper through its application in partner projects. In the following sections, we illustrate the process with the case of the Jaguars of Iguaçu Project, the first partner project to adopt the approach. The Jaguars of Iguaçu Project[1] was created in 2018 and has subsequently carried out jaguar population surveys and a variety of outreach activities including technical assistance to ranchers, community engagement, education and communication. The project has also conducted a social survey in which 85 ranchers were interviewed. The results from this survey

[1]https://www.oncasdoiguacu.org/

supported some of the assumptions underpinning the theory of change described below.

Our workshop process was first applied in October 2019 in a two-day pilot in-person workshop conducted in the administrative office of PNI. The workshop was facilitated by a representative from the academic sector, and the five project team members and two park staff representatives participated in the workshop. This pilot workshop was also attended by five representatives of the Yaguarete Project (created in 2002). Both Jaguars of Iguaçu and Yaguarete projects are dedicated to the conservation of jaguars, concentrate their actions in complementary, adjacent areas in the Upper Parana Region–PNI in Brazil and Corrientes region in Argentina, respectively–and have continuous collaboration in research. The Upper Parana Region is part of the Atlantic Forest and one of the most critical areas for jaguar conservation. Three key protected areas–Iguaçu National Park and Turvo State Park in Brazil, and Iguazu National Park in Argentina–host an estimated population of 100 jaguars, representing one-third of all jaguars in the Atlantic Forest (Morato et al., 2013).

The goal of the first workshop was to introduce the process to the participants, exposing them to each of the key steps: situation assessment and goal setting, system mapping and identification of leverage points, and production of a ToC and of a framework for monitoring and evaluation (M&E). The outcomes of the pilot workshop, with a focus on the ToC and the M&E framework, were further developed and refined in two follow-up online meetings with the project's team in 2020 and early 2021.

WHERE WE ARE AND WHERE WE WANT TO GET: A CONCEPTUAL FRAMEWORK

Human-wildlife coexistence has drawn increasing attention from researchers (Frank et al., 2019; Koenig et al., 2020; Pooley et al., 2021). Although relatively new to the HWI literature, the concept of coexistence has been addressed from multiple perspectives, with an emphasis on its conceptualization (Carter and Linnell, 2016; Pooley et al., 2021), relationship with similar concepts–tolerance and acceptance–and with HWC (Frank, 2016; Glikman et al., 2021), and scales and levels of analysis (Carter et al., 2019; Koenig et al., 2020; Pooley et al., 2021). The framework described below is intended to contribute to this increasing body of knowledge by providing an approach for placing HWI in the context of planning and management concern.

Expanding from the concept of conflict-to-coexistence continuum (Frank, 2016; Frank et al., 2019), which suggests a one-dimensional graphical representation to depict HWI in its range from mutually negative to mutually positive outcomes, we propose that HWIs be described by their two dimensions–wildlife and human situation–separately (**Figure 1A**). Each situation is typically informed in terms of population-level parameters such as population size or conservation status (e.g., IUCN conservation categories), whereas human situation, in the context of the interaction, has been expressed in terms of both tangible and intangible factors such as financial cost/benefit, attitude (i.e., favor/disfavor), feeling (e.g., like/dislike), and wellbeing. The framework can also accommodate individual-level parameters such as animal welfare, which are increasingly considered in wildlife conservation and management (Sekar and Shiller, 2020). In the context of planning, the choice of the parameter is based on feasibility besides relevance. For instance, individual human wellbeing is arguably what ultimately matters, but its measurement can be challenging, so decision makers may select another parameter that, while also relevant, can be objectively assessed (e.g., attitude) so that management results can be tracked and demonstrated. For decision-making purposes, each axis can inform a single parameter or a set of parameters, one at a time or combined (e.g., in an index).

The wildlife and human axis combined define four archetypical representations (Fischer et al., 2017; Hartel et al., 2018) that cover all possible HWIs: (i) negative for both wildlife and people (left lower quadrant), like when endangered wildlife causes damage to people and preventive or retaliatory killing or harassment ensues (e.g., Das and Jana, 2018; LaDue et al., 2021; Simpfendorfer et al., 2021); (ii) negative for wildlife and positive for (some) people (left upper quadrant), as in overharvest associated with poaching or wildlife trade (e.g., Shepherd et al., 2017; Gomez et al., 2020); (iii) positive for wildlife (at the population level) and negative for people (right lower quadrant), as when abundant wildlife is a nuisance (e.g., Gamalo et al., 2019; Carpio et al., 2021); vehicle collisions and zoonotic diseases produce negative outcomes to people and are associated with both endangered and abundant wildlife (e.g., Pagany, 2020; Namusisi et al., 2021), therefore they belong in the two lower quadrants; and (iv) positive for both wildlife and people (right upper quadrant), like when abundant, native or exotic wildlife, is used in tourism (e.g., Macdonald et al., 2017) or sustainable harvest (e.g., Campos-Silva et al., 2017) (both, but most obviously the latter, may only apply to population-level parameters and not to individual-level).

In Latin America, where the approach described in this paper has been developed and applied, the Spanish and Portuguese word used to illustrate the condition in the right upper quadrant of the framework is *convivencia/convivência* which means, literally, to live together. *Convivencia* has a positive connotation. It is not only about sharing the space (as in co-occurrence or cohabitation), but also mutually benefiting from the interaction, even if the benefit is intangible (e.g., enjoying the presence of each other). The goal of HWC management is to "shift" situations in the left and lower half of the framework to the right and up, respectively, toward *convivencia*. Yet generally desirable, the win-win condition implied by *convivencia* is seldom realistic (Vucetich et al., 2018) and not ultimately necessary for conservation. Instead, in many instances it may be good enough to achieve *a condition whereby none of the parties involved–wildlife and human–receives a significant negative impact from the other, so that their "existing together" can be sustained* (**Figure 1A**). This is the operational definition of coexistence adopted in this approach, and the ultimate goal of planning for human-wildlife coexistence would be to move HWIs in the left and bottom halves to the right and upwards until coexistence is reached. As a note, *convivencia* implies coexistence (a mutually beneficial interaction can only occur when the parties

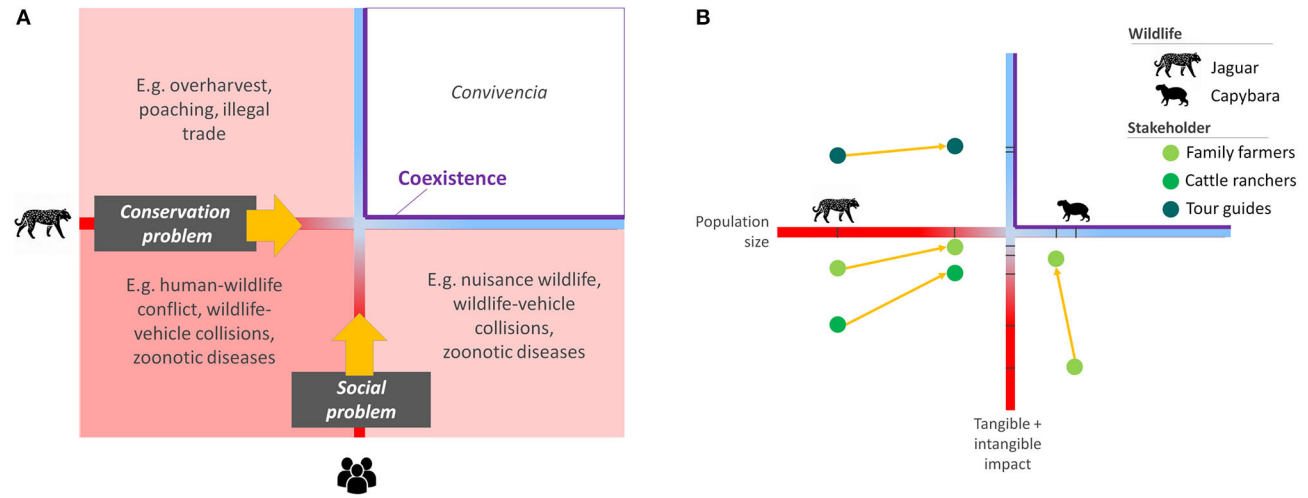


FIGURE 1 | Conceptual framework for describing human-wildlife interactions based on wildlife and human situations separately. **(A)** Situation ranges from poor (red) to good (blue). Parameters for wildlife and human situation are chosen according to relevance and accessibility (e.g., species population size and perceived impact, respectively). Arrows inform the direction of desired change from the conservation and social perspectives. Coexistence is defined as the condition whereby none of the parties involved–wildlife and humans–receives a significant negative impact from the other (purple line), so that their "existing together" can be sustained; **(B)** Hypothetical use of the framework to set goals and communicate current and desired situation in planning for multi-species (e.g., jaguar and capybara) and multi-stakeholder (e.g., family farmers, cattle ranchers and tour guides) coexistence. In this hypothetical but realistic example, the objective is to improve the situation of the parties involved toward coexistence, while recognizing that coexistence will not be fully achieved with available resources (e.g., time).

involved exist together) but not necessarily the other way around, as two parties can coexist without a win-win interaction (they can coexist even with some degree of conflict!).

This two-dimensional framework for describing HWI can be used to "map" any HWIs of management interest (**Figure 1B**), also allowing for multiple stakeholders and species. In the context of planning, the framework is used to visually inform both the current situation and desired changes within a specific timeframe (**Figure 1B**). When multiple stakeholders are portrayed, their current and desired situation can reveal actual and potential common ground and conflict. Such graphical representation can be a particularly useful tool for goal-setting and of communication in stakeholder engagement processes.

The framework also provides objective criteria for clarifying confusing terminology such as coexistence and *convivencia* (or equivalent). More than just a matter of semantics, these two terms refer to fundamentally different goals in HWI management. Other terms that have been used interchangeably with coexistence and *convivencia* are co-occurrence and cohabitation. Co-occurrence and cohabitation refer to the necessary ecological condition for any HWI to happen: the two species coincide in space and time, regardless of the outcome from their interaction (Waldron et al., 2013). In all HWIs depicted in the four quadrants of the framework humans and wildlife co-occur or cohabitate (the later term arguably connoting greater proximity between the parties).

Furthermore, the graphic representation makes a clear distinction between the conservation and social dimensions of a HWI problem (left and lower halves of the figure, respectively), encouraging decision-makers and managers to explicitly address each of them. When an endangered species does not cause any significant perceived impact on people, i.e., the interaction does not have an important social dimension, the situation and desired change can be properly expressed unidimensionally along the wildlife situation axis: it is about conservation only. But whenever HWIs have social implications, either negative or positive, situation assessment, and therefore planning, will benefit from such a framework that integrates the ecological and social dimensions.

In the workshop, participants were asked the following questions (Q1–Q6), whose answers (A1–A6) served to populate the framework: Q1. *What changes are intended to be caused?* A1. To improve the situation of both sides of the human-jaguar relationships; Q2. *What parameters are used to describe this change?* A2. Jaguar population size and local attitude toward jaguar conservation (other parameters were used but for the sake of illustration, we focus on attitude in this paper); Q3. *What are the target social groups?* A3. Family farmers; Q4. *What is the magnitude of the change?* A4. From the current 28 to 50 jaguars, and from 75 to 95% of the farmers favorable (as opposed to unfavorable) to jaguar conservation; Q5. *Where is the change expected to happen?* A5. In the 14 municipalities adjacent to Iguaçu National Park, home to approximately 500,000 people; and Q6. *When is the change expected to happen?* A6. Within 5 years.

These questions were not resolved sequentially but iteratively. The answer to one question can affect the answer to other questions. For example, the lack of a proper baseline (Q4) may result in the need to revisit the proposed parameters (Q2), and the magnitude of the desired change (Q4) may determine the expected timeframe (Q6). A cornerstone of planning for human-wildlife coexistence is that a project must be able to demonstrate

its success, hence the importance of selecting the right (i.e., relevant and viable) parameters, indicators, and timeframe. The exercise is useful for getting all participants on the same page regarding the issue they are addressing and the changes they want to cause. The resulting graphic display describes in a nutshell the *what, who, how much, where*, and *when* of their particular project. The next question to be addressed is, then, *how* to cause the desired change.

HOW TO GET THERE–AND SHOW THAT YOU DID IT: SYSTEMS THINKING AND THEORY OF CHANGE

A key assumption in the proposed change-focused approach is that HWIs are embedded in a system (i.e., "a group of interacting or interrelated elements that act according to a set of rules to form a unified whole", Merriam-Webster, 2019). Actually, HWI issues typically involve interacting ecological, economic, and sociopolitical elements, with complex and adaptive dynamic relationships driven by the thoughts, feelings, and power of the associated actors (Bunnefeld et al., 2017). Therefore, systems thinking, defined as "a set of synergistic analytic skills used to improve the capability of identifying and understanding systems, predicting their behaviors, and devising modifications to them in order to produce desired effects" (Arnold and Wade, 2015), can be useful for examining HWIs. Systems approaches emphasize the need to understand dynamic interrelations between various components (Von Bertalanffy, 1993), shifting the emphasis from isolating the causal effect of a single factor to comprehending the functioning of the system as a whole. Feedback loops and unintended effects, in addition to linear chains of causality, are central to systems thinking. In application, systems thinking has been defined as both a skill and an awareness (Sanko et al., 2021).

In the workshop, systems thinking is used to develop a system map depicting in terms of "nodes and connections" the factors that determine the change. The emphasis is on whether the boundaries and interrelationships of the proposed system accurately reflect the story the group is trying to depict. System maps are shorthand descriptions of what we perceive as current reality. If they reflect that perspective, they are "right enough." Proper facilitation techniques are used to help participants analyze the map and draw conclusions on potential leverage points i.e., where interventions could bring about the desired changes in a more cost-effective way (Meadows, 1999; Abson et al., 2017). If all the causes cannot be overcome by the project, it is important to prioritize the pathways to intervene. The participants then select one or more solution pathways to form their strategy, based on explicit criteria such as project's objectives and priorities, preferences of key stakeholders, cost-effectiveness and technical feasibility.

Once the system is understood and the leverage points for each selected causal pathway are identified, the next step in the workshop process is to describe in detail how the change is expected to happen. In other words, it is time to create a Theory of Change (ToC). A ToC is a decision support tool that illustrates the causal links and sequences of events needed for an activity or intervention to lead to a desired outcome or impact (Center for Theory of Change, 2013). It is both a process and a product (Vogel, 2012). Fundamentally, the participants describe the causal pathways in terms of inputs, activities, outputs (products), short- to long-term outcomes (effects) and desired final impact, choose indicators for each product and effect and, in doing so, generate a framework for monitoring and evaluating results. Intermediate outcomes must be clearly articulated within the ToC. This is perhaps the most important part of the process: too often project teams jump from their activities to their final goals without thinking through the changes that need to happen in between. Indeed, the process of creating a ToC enables a better understanding of the underlying assumptions and questioning of the assumptions that are often side-lined, in the specific context where activities and interventions take place (e.g., electric fences cause less livestock depredation, which causes higher tolerance, which causes less persecution). This can help to identify knowledge gaps and guide research, as an additional benefit from producing a ToC.

A major advantage of this approach–systems thinking followed by ToC–is that the context analysis and decision-making are integrated. Traditional ToC diagrams usually depict only the actions that a particular organization or program plans to implement, together with the related changes they anticipate through the implementation of those actions. Organizations imply that positive change (e.g., increased wildlife population) results directly and solely from their actions, rather than from a range of interrelated contextual factors, of which their program is part. Starting with a system map and integrating a ToC can be an effective way to address this issue.

As a conservation-oriented project, Jaguars of Iguaçu ultimately aims at increasing jaguar population size up to a viable and sustainable level, while improving the actual and perceived impact of jaguars and of the INP on local communities. In order to encourage the workshop participants to consider from local to distant causes, the systems approach to examine the factors that determine change in jaguar population size was structured in different levels of analysis: ecological, (human) behavioral, personal, social/institutional, and societal (**Figure 2**). System mapping started with the ecological factors that directly determine jaguar population size: mortality, natality, immigration and emigration. These in turn are affected by changes in habitat quality and prey base besides human behavior: intentional and unintentional killing of jaguars, intentional and unintentional killing of prey, changes in land use and habitat management. Participants were then briefed on some of the main conceptual frameworks that have been used to explain and predict human behavior such as the Theory of Planned Behavior (Fishbein and Ajzen, 2010) and Hazard Acceptance Model (Bruskotter and Wilson, 2014). The explanatory variables include attitudes, perceived social norms, and perceived behavioral control regarding jaguar killing; tolerance to jaguars, which in turn is determined by perceived costs and benefits, affect toward the species, and trust in the management agency; motivations vs. perceived barriers; and level of awareness, knowledge and skills. Factors at the social and institutional level include the level of engagement, the magnitude of economic incentives, technical

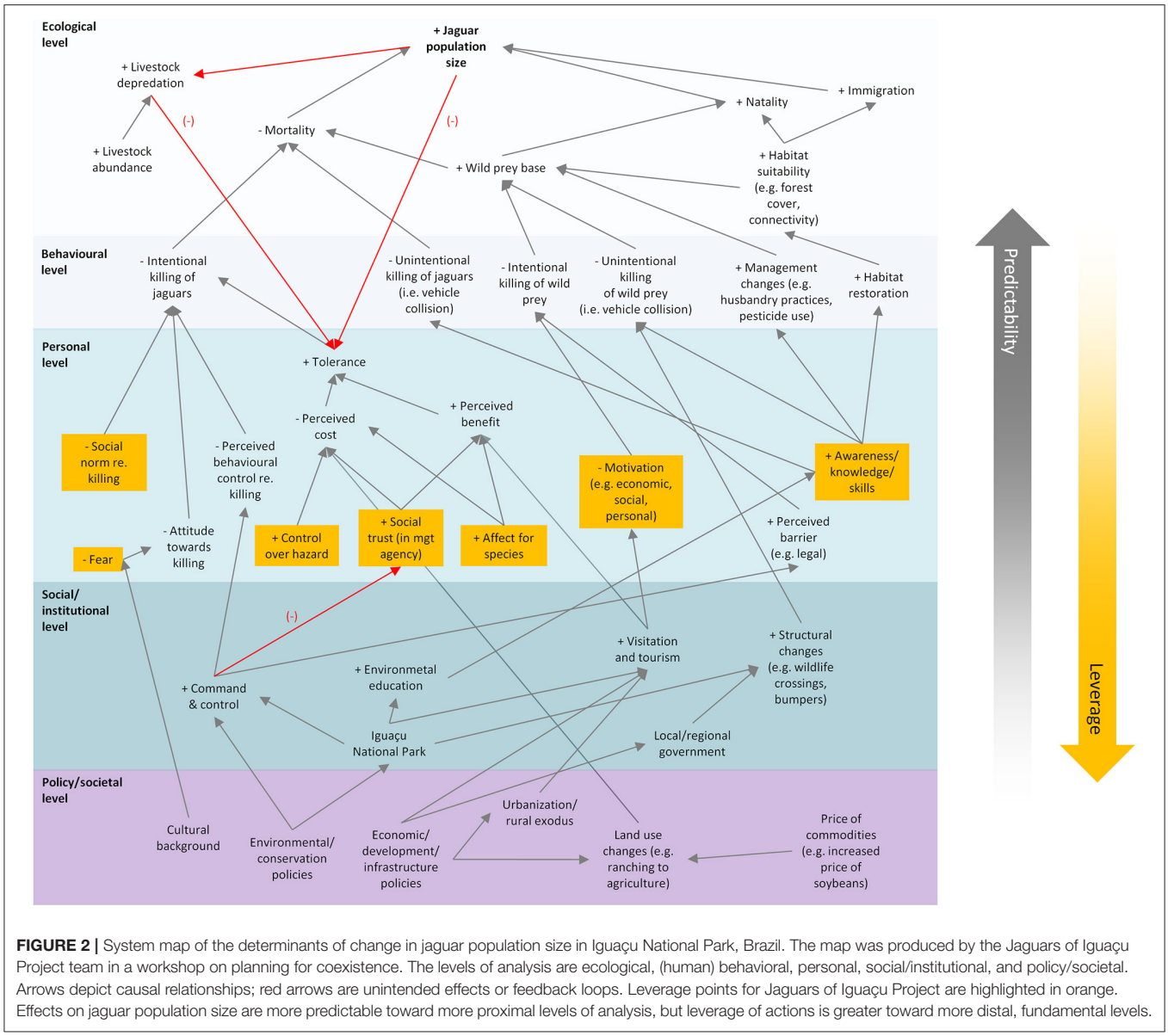


FIGURE 2 | System map of the determinants of change in jaguar population size in Iguaçu National Park, Brazil. The map was produced by the Jaguars of Iguaçu Project team in a workshop on planning for coexistence. The levels of analysis are ecological, (human) behavioral, personal, social/institutional, and policy/societal. Arrows depict causal relationships; red arrows are unintended effects or feedback loops. Leverage points for Jaguars of Iguaçu Project are highlighted in orange. Effects on jaguar population size are more predictable toward more proximal levels of analysis, but leverage of actions is greater toward more distal, fundamental levels.

assistance in agriculture, and of command and control, and the role of protected areas, all determined by institutional capacity and financial resources. At the societal level, urbanization, changes in land use, the growth of tourism, and the national action plan for the conservation of large felids were added to the map.

Throughout the analysis, systems thinking allows participants to see previously hidden linkages, including unanticipated side effects—e.g. the negative impact of protection on people's trust in the park authority—and feedback loops, like the negative effect of more jaguars on people's tolerance to jaguars. In the HWI literature, the factors that more proximally and directly determine the situation, at the upper levels of analysis in **Figure 2**, have received more attention. However, the large-scale and sustained condition implied by coexistence–as opposed to a temporary truce–requires a more in depth understanding of the system, which is achieved by addressing the factors at the social, institutional and societal levels of analysis (Massarella et al., 2021). Besides, while it is generally easier to detect the effects of actions implemented at the ecological, behavioral and personal factors, the more fundamental the level of intervention, the higher the leverage.

For the development of a ToC for Jaguars of Iguaçu Project, specific pathways were selected, taking into account the desired impact of the project, the activities already underway, the databases available, and the feasibility of collecting additional data to serve as indicators of intermediate outcomes. For each pathway, a detailed results chain was articulated connecting activity to outputs to outcomes to impact. This information was organized in a logical framework, informing also the respective

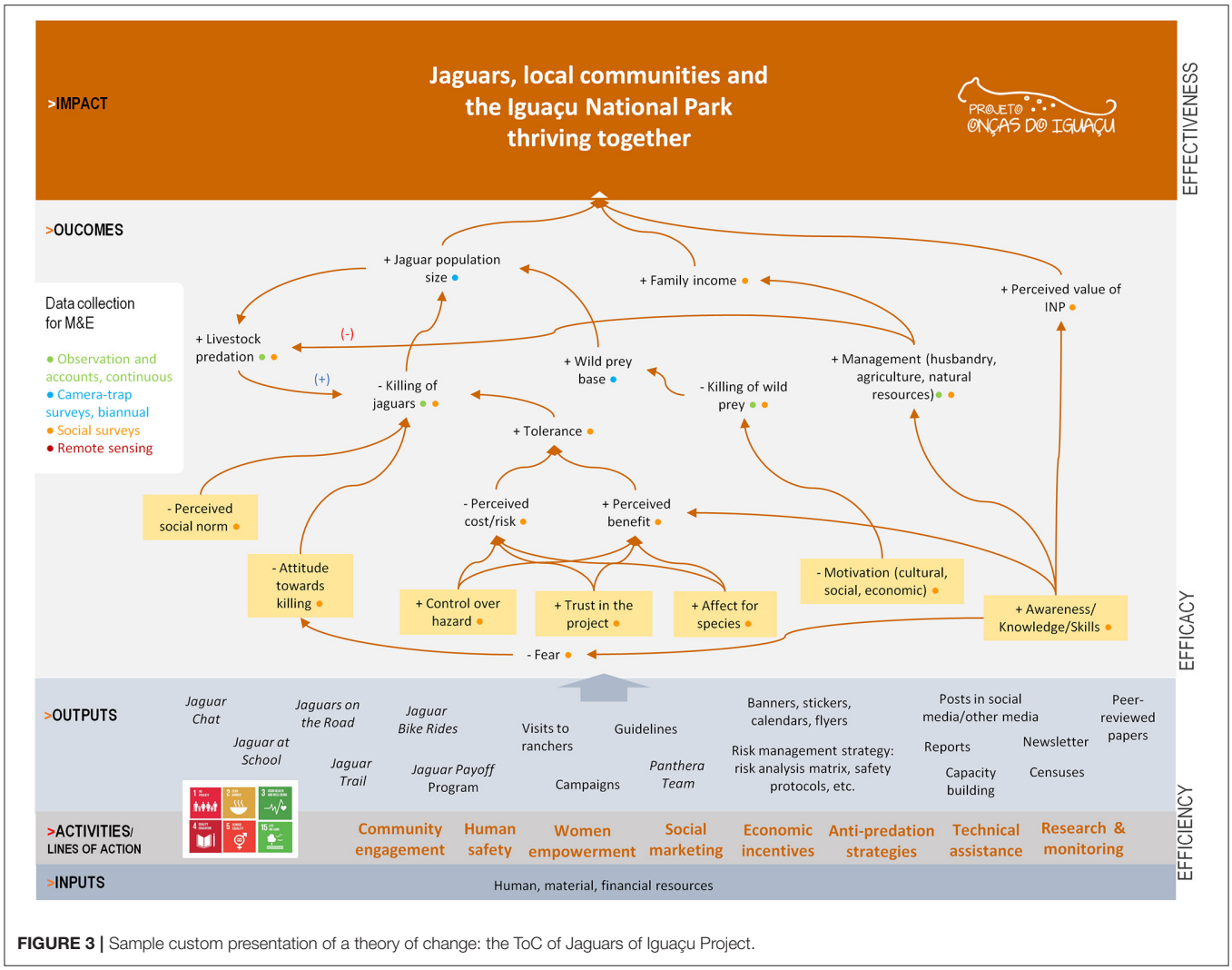


FIGURE 3 | Sample custom presentation of a theory of change: the ToC of Jaguars of Iguaçu Project.

indicators, means of verification, and assumptions behind each causal link. Next, the project team informed baseline values and agreed upon target values for each outcome, the timeframe to achieve the target values, and the estimated budget to do it. Altogether, this dataset is a framework for evaluating the effectiveness of the project (i.e., extent to which the desired impact is caused), the efficacy of the actions (i.e., extent to which planned short- and long-term outcomes are achieved), and the efficiency of the project and each action (i.e., the ratio of outputs to inputs in terms of time, energy, and money). Custom decorations were used to make the project's ToC look attractive (**Figure 3**). While this sounds superficial, it can be useful to make those all-important presentations to donors, board members and key stakeholders.

In addition to the product summarized in a logic model or results chain, the ToC is a process that gives organization and program teams the opportunity to think, discuss, learn from each other, collaborate, and develop a sense of ownership of the process. It strengthens projects through more considered decision-making and stronger teams as people are brought together. It also enables projects to identify knowledge or capacity gaps as they appear and facilitates projects to evolve and become refined over time through adaptive management.

Future Directions: Advancing a Transdisciplinary Model for Planning

The planning and managerial perspective in which human-wildlife interactions is discussed above has an explicit emphasis on change. More specifically, it is about changing HWI toward the benefit of both wildlife and people. Accordingly, delivering change should be the primary focus of research for human-wildlife coexistence. The evidence on which decision-making is based must come not only from research on wildlife and on people, but preferably also from research on how the system changes in response to management actions. Such changes obligatorily affect people, and the associated costs and benefits are not always distributed equally among interest groups or over time (e.g., for some groups the long-term benefit may imply short-term costs). Therefore, creating the conditions for these groups to participate in decision-making is a moral imperative and

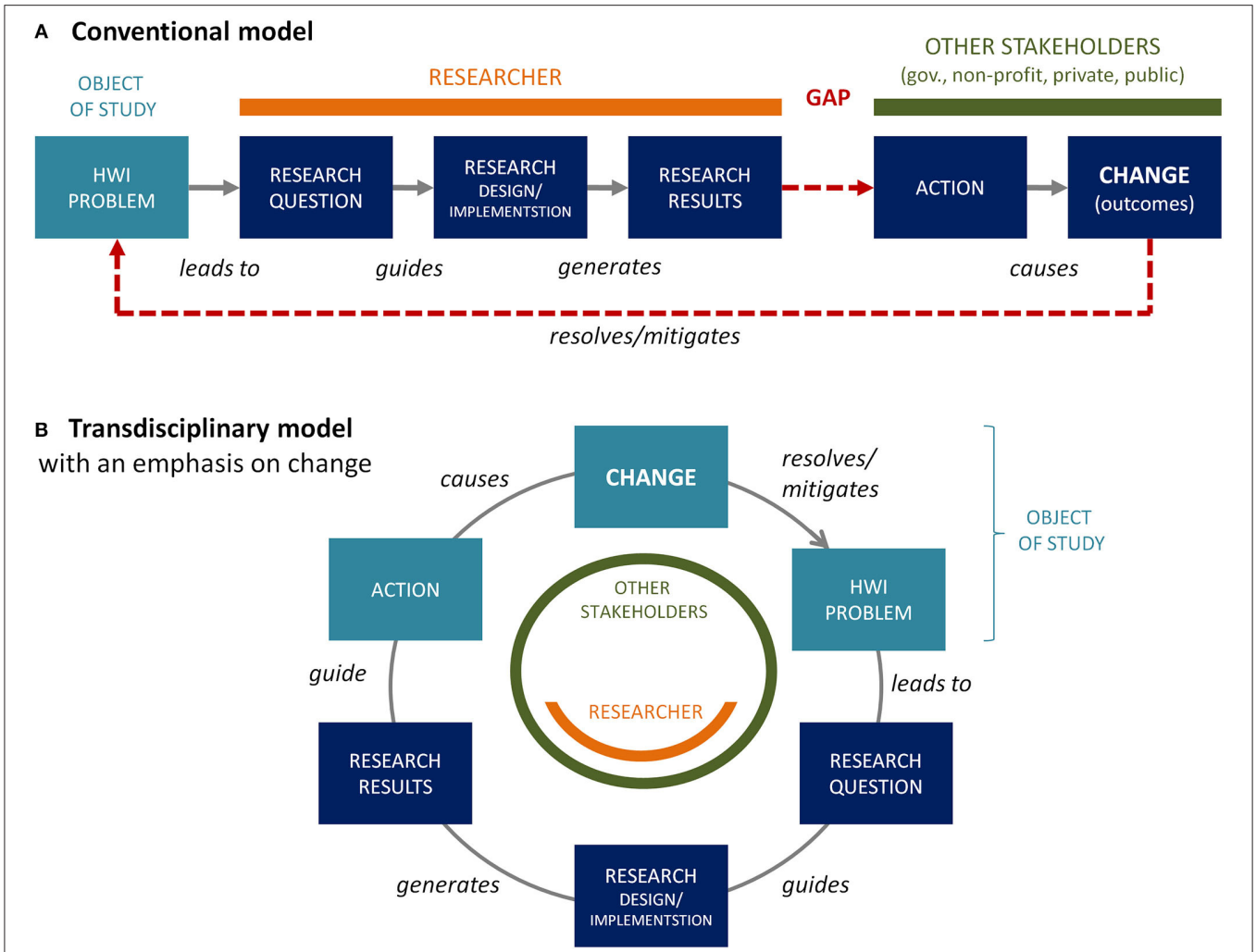


FIGURE 4 | Conventional **(A)** and alternative **(B)** models for the role of researchers and other stakeholders in tackling HWI problems. Dashed arrows represent the links that are typically weak or missing: the use of research results by decision makers and practitioners, and the monitoring and evaluation of the ultimate effects of actions on the HWI problem.

also more promising, complementing top-down approaches that might, on their own, result in lack of buy-in and implementation (Treves et al., 2009; Dietsch et al., 2021; Salvatori et al., 2021; Vucetich et al., 2021b).

The proposed process of planning for human-wildlife coexistence favors an alternative to the conventional model of research-implementation in HWI in which a gap can too often separate academia from decision-makers and other stakeholders (**Figure 4A**). Actually, the science-policy gap is not just a concern in conservation practice, but an urgent challenge to be addressed in many fields (e.g., climate change). Most academic research in HWI has focused on describing and explaining HWC-related problems. Levels of livestock loss to predators and of retaliatory killing, and the factors determining these phenomena, are examples of objects of such research (e.g., Bruskotter et al., 2017; Chaka et al., 2021, respectively). Research objects are chosen according to academic and scientific criteria, including the adequacy and feasibility of the research within the norms and timeframes imposed by graduate programs and funding bodies. Researchers' personal interests and preferences, and scholarly originality, also play a role in the selection of research topics. In this model, the contribution of academia can end with the publication of research results in scientific journals, typically in academic language and in English, regardless of the language spoken by the stakeholders–decision-makers, managers, and residents–of the study site/system. These actors, in turn, have made limited use, if any, of the results of scientific research, whether because of difficulties in understanding them, limitations in accessing them, or being overwhelmed with information and studies, some of which contradict each other (Wesselink and Hoppe, 2020). Without a robust evidence base, teams in charge of projects and programs measure the success of their activities based on outputs and short-term outcomes, at best, but the connection between

their results and the impact on the HWI problem at hand is often not demonstrated.

Planning for coexistence will benefit from a process that differs from the conventional model in three major points (**Figure 4B**): (i) research objectives go beyond describing and explaining the problem to address also how the problem is resolved or mitigated by management actions and their outcomes, (ii) decision-makers, managers and residents participate in all stages of the process: research questions, for example, are not chosen only for their academic and scientific merit, but mainly for their relevance to these stakeholders, i.e., the contribution of academia is directly guided by concrete demands of specific stakeholders; and (iii) the process is explicitly cyclical and iterative, and the emphasis is not on a definitive solution–which in fact rarely exists–but on adaptation and resilience.

This transdisciplinary model with a focus on adaptive change constitutes an approach for creating the knowledge, skills, and collaborations necessary among researchers, practitioners and stakeholders for furthering human-wildlife coexistence. The transdisciplinary approach, by definition, integrates fields beyond academia with academic research, and engages stakeholders in knowledge co-production, through processes of collective inquiry and reflection with relevant stakeholders (Lang et al., 2012) that foster ownership and full participation. Transdisciplinarity has indeed been increasingly mentioned as a promising way of producing knowledge and decision-making in the context of the world's most pressing issues (Macdonald, 2019; Rigolot, 2020). Nonetheless, despite the growing interest in transdisciplinary approaches among sustainability scientists and practitioners (Sharpe et al., 2016; Rocha et al., 2020), their use in the field of HWI is still in its infancy (Hartel et al., 2019; Jiren et al., 2021).

The planning for coexistence workshop with Jaguars of Iguaçu Project integrated academia (University of São Paulo) with the government (Iguaçu National Park) and non-profit (the project itself) sectors as the starting point of a continued and adaptive process. Local stakeholders' needs have guided scientific research, and research results have subsidized the design and implementation of interventions. The mapping of the stakeholders of jaguar conservation, both in the Iguaçu region specifically and in the Atlantic Forest as a whole, was done in a separate workshop, as part of a partner project of the Jaguars of Iguaçu project. Stakeholder analyses were used to identify the stakeholders and group them according to their levels of participation, interest, and influence in the project, and to determine how best to involve and communicate each of these stakeholder groups throughout (Sandroni et al., submitted). Local stakeholders, however, have not participated in-person in the planning workshop. Their needs and aspirations have been assessed through surveys and taken into account in the process. A challenge ahead facing planning for coexistence will be to implement and refine mechanisms for greater stakeholder participation (Vucetich et al., 2021b)—local farmers and ranchers, in the case of jaguar conservation—ensuring that the transdisciplinary model proposed here is fully implemented.

CONCLUSIONS

The process outlined here provides a generally usable template for how to conduct evidence-based, structured, and participatory planning for human-wildlife coexistence. We hope it can help to overcome a major stumbling block in the transformation of problematic HWI into coexistence i.e., the vagueness of goal and pathway. While we are still far from generating a predictive theory of coexistence, current efforts to improve HWI can benefit from more systematic and inclusive ways of making decisions. Naturally, as research findings reveal the high degree of complexity and local specificity of human-wildlife and human-human interactions (Zimmermann et al., 2021), the specific methodological steps of the proposed approach need to be adjusted according to the study area, stakeholders involved, and resources available. Ready-made and one-size-fits-all solutions for HWI problems are scarce, hence the potential benefit of our planning approach.

The process of planning for coexistence as proposed here can complement current approaches such as threats analyses and action plans which, as the names suggest, place relatively more emphasis on threats and actions than on results i.e., change. In addition, it can expand the reach of workshop processes, analytical tools, and monitoring and evaluation frameworks currently in use (e.g., Open Standards, the tools of the IUCN SSC Conservation Planning Specialist Group) and especially the scope of HWI management and policies, traditionally concentrated on negative interactions involving threatened species.

Given the growing importance of coexistence and the associated more holistic, fairer ways of addressing HWIs, the approach outlined here has great potential for tackling current and future pressing HWI issues. The realization of this potential, however, will depend on a greater support from funding bodies for long-term, interdisciplinary, collaborative research focusing on change and on ways to monitor and evaluate results. It is important to make training in decision-making and solutions-oriented, actionable science more accessible in academic and informal learning environments. Also, mechanisms for data sharing and collaboration involving researchers, government agencies, non-profit organizations, and the private sector will need to be improved.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

SM wrote the first draft. KF, AZ, and DM contributed to the conception of the study and contributed revisions. VF and YB contributed revisions and together with AK and TR collected data in the planning workshop.

All authors contributed to the article and approved the submitted version.

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Fostering Coexistence Between People and Large Carnivores in Africa: Using a Theory of Change to Identify Pathways to Impact and Their Underlying Assumptions

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Coexistence with large carnivores poses challenges to human well-being, livelihoods, development, resource management, and policy. Even where people and carnivores have historically coexisted, traditional patterns of behavior toward large carnivores may be disrupted by wider processes of economic, social, political, and climate change. Conservation interventions have typically focused on changing behaviors of those living alongside large carnivores to promote sustainable practices. While these interventions remain important, their success is inextricably linked to broader socio-political contexts, including natural resource governance and equitable distribution of conservation-linked costs and benefits. In this context we propose a Theory of Change to identify logical pathways of action through which coexistence with large carnivores can be enhanced. We focus on Africa's dryland landscapes, known for their diverse guild of large carnivores that remain relatively widespread across the continent. We review the literature to understand coexistence and its challenges; explain our Theory of Change, including expected outcomes and pathways to impact; and discuss how our model could be implemented and operationalized. Our analysis draws on the experience of coauthors, who are scientists and practitioners, and on literature from conservation, political ecology, and anthropology to explore the challenges, local realities, and place-based conditions under which expected outcomes succeed or fail. Three pathways to impact were identified: (a) putting in place good governance harmonized across geographic scales; (b) addressing coexistence at the landscape level; and (c) reducing costs

and increasing benefits of sharing a landscape with large carnivores. Coordinated conservation across the extensive, and potentially transboundary, landscapes needed by large carnivores requires harmonization of top-down approaches with bottom-up community-based conservation. We propose adaptive co-management approaches combined with processes for active community engagement and informed consent as useful dynamic mechanisms for navigating through this contested space, while enabling adaptation to climate change. Success depends on strengthening underlying enabling conditions, including governance, capacity, local empowerment, effective monitoring, and sustainable financial support. Implementing the Theory of Change requires ongoing monitoring and evaluation to inform adaptation and build confidence in the model. Overall, the model provides a flexible and practical framework that can be adapted to dynamic local socio-ecological contexts.

Keywords: large carnivore conservation, African semi-arid, community-based conservation, human wildlife conflict, community-based natural resource management, adaptive co-management, rangeland management, climate change adaptation

INTRODUCTION

As the global human population, accompanied by rapidly rising per capita consumption, climbs toward 10 billion (Crist et al., 2017; United Nations, 2017), the intensifying impacts of climate change and environmental degradation pose an increasing threat to global biodiversity (IPCC, 2014). Africa, with a projected doubling of its current population over the next three decades (United Nations, 2017), faces particularly acute pressures on its natural resources in the near future. It is also a continent that has already been heavily impacted by climate change, including a higher frequency and intensity of droughts, increased desertification, reduced rangeland productivity, and heightened food insecurity (IPCC, 2019). These impacts are predicted to intensify over the coming years as the planet continues to warm, and pressures on natural resources increase (Shukla et al., 2019). Mitigating against the consequent impacts on biodiversity will require transformative change that supports the sustainable coexistence of people and wildlife, while increasing resilience and contributing to the development of rural communities.

In this context, Africa's large carnivores present both challenges and opportunities for navigating through contentious and often opposing demands on land, biodiversity, and natural resource extraction. In the face of Africa's rapidly growing human population, setting aside additional protected areas that exclude human activities may raise insurmountable challenges for many vulnerable and marginalized rural communities, who are often dependent on natural resources for their livelihoods. Yet, a substantial proportion of the distributional range of Africa's large carnivores [e.g., 78% of cheetah *Acinonyx jubatus* (Durant et al., 2017) and 83% of leopard *Panthera pardus* (Jacobson et al., 2016)] is outside current protected areas in mixed-use landscapes. Outside protected areas large carnivores face increasing and multiple threats, including conflicts due to livestock depredation, loss of prey and habitat, and land degradation and fragmentation (Ripple et al., 2016). However, large carnivore presence also indicates alternative possibilities for the management of multiple-use landscapes, if wildlife can provide value to local communities. Ultimately, the continued survival of large carnivores will depend on long-term support for their conservation and on the tolerance of communities who share their landscapes.

What Do We Mean by Coexistence?

A myriad of interactions between people and wildlife may occur when communities share their land with wild animals. Although coexistence generally describes situations when these human-wildlife interactions result in sustainable wildlife populations (Phalan et al., 2011), our understanding of coexistence does not exclude the presence of conflict, since an expectation of rural people to develop overwhelmingly positive attitudes toward carnivores and to share a landscape with them without incurring conflict is unrealistic (Linnell, 2013). Indeed, peoples' relationships with wild carnivores are rarely static or constant, but encompass multiple emotions including fear, admiration, reverence, or anger, sometimes even simultaneously (Bhatia et al., 2021). Moreover, climate change is expected to modify relationships between people and large carnivores, often exacerbating conflicts (Abrahms, 2021), but may also support coexistence in some areas. For example, in the forests around Golestan National Park in Iran, declines in humidity have reduced disease outbreaks which have, in turn, mediated a reduction in conflict between livestock keepers and leopards (Khorozyan et al., 2015).

We seek to understand coexistence dynamically and holistically, including positive aspects of human-wildlife relationships, alongside the more widely publicized negative interactions such as crop damage, livestock depredations, attacks on humans, and retaliatory killing (Pooley, 2021). We therefore accept coexistence as "a state where conflict exists but where interactions are kept within acceptable limits" (Linnell, 2013, p. 26). This is a characterization of coexistence as a dynamic state in which interactions between people and carnivores can

be governed by diverse institutions to ensure the sustainability of carnivore populations, social legitimacy, and tolerable levels of risk (Carter and Linnell, 2016). Thus, this definition has the flexibility to encompass the politics that govern both the interactions between people and carnivores and the relations between people with competing interests concerning carnivores (Redpath et al., 2013; Carter and Linnell, 2016; Jepson et al., 2018). Because this definition is dynamic, it can also encompass changing environmental states, such as may result from climate change (Abrahms, 2021).

Multiple ethnographic studies have examined coexistence from the perspectives of local communities demonstrating the complex, and often ambivalent, ways in which local people establish relationships with the natural world, via their livelihoods, cultures, lived experiences and everyday practices. For example, research by Pooley (2016) has documented nuanced and varied human relationships with crocodiles across African geography and history; Baynes-Rock (2013) and Gebresenbet et al. (2018) describe cultural beliefs that bring communities in Ethiopia to view hyenas as beneficial and reasonable beings, despite high rates of livestock depredation, and attacks on humans; whilst Goldman et al. (2010) document the ways in which superficially negative relationships between Maasai and lions, rooted in conflict (Ikanda and Packer, 2008), conceal the role that ritual lion killing plays in providing the cultural underpinning of powerful feelings of respect and admiration for lions. Approaches that build on such deep cultural relationships with large carnivores can play fundamental roles in promoting the value of carnivores as a social as well as a natural resource (Nijhawan and Mihu, 2020). These examples contribute to an understanding of coexistence as complex, multi-layered and deeply rooted in culture, and demonstrate the importance of viewing coexistence through different perspectives and cultural lenses (Adams and Mulligan, 2003; Peterson et al., 2010; Pooley, 2021).

The relationships between people and wildlife are also impacted by external political and economic processes, which influence the shape of conservation interventions and their social and economic impacts on local communities. There has been substantial research revealing the impacts of centrally imposed protected areas on local communities, including land dispossession, community displacement, and livelihood disruption (Igoe, 2006; West et al., 2006). Other research has looked at the ways in which financial instruments, intended to offset the costs of coexistence, have reconfigured human-animal relations (Nyhus et al., 2005; Fletcher, 2010); the effects of tourism businesses on local communities (Bluwstein, 2017; Homewood, 2017); and the disruption of traditional and cultural practices due to policies around community-based natural resource management (Nelson and Agrawal, 2008). This growing body of research exposes how political negotiations and decisions can shape coexistence, and demonstrates how conservation and development interventions may have unintended impacts due to the complex ways in which they are mediated by local cultures, and historical and contemporary power dynamics.

International conservation paradigms may also mediate local relationships with nature and experiences of coexistence (Robbins, 2012). From the turn of the century, conservation has been dominated by a utilitarian approach to nature requiring a careful evaluation of the economic and material costs and benefits of coexistence within an ecosystem services framework (Mace, 2014). However, more recently, there has been a shift to a more nuanced understanding of the two-way relationships between people and nature, incorporating less tangible and more multifaceted components of well-being that constitute a "good life" and shape socio-ecological relations (Woodhouse et al., 2017; Pascual et al., 2021). Here, well-being is conceptualized across three main dimensions: objective material needs; subjective meaning and satisfaction, including feelings of value, fairness, and change; and social needs, including people's ability to fulfill social obligations and conventions by pursuing, for example, livelihoods that contribute to people's sense of identity and way of life (Chan et al., 2016; Woodhouse et al., 2017). The inclusion of social and subjective components of well-being, in addition to material components, enables the accommodation of diverse needs and aspirations within communities across different gender, age, ethnicity, class, and livelihood groups. This multidimensional approach to well-being has been incorporated into the "nature's contribution to people" discourse of the Intergovernmental Panel on Biodiversity and Ecosystem Services (Diaz et al., 2018) and facilitates a deeper understanding of the quality and local experience of coexistence with wildlife.

In this article we use a Theory of Change approach to identify logical pathways that can promote and improve the multidimensional experience of coexistence of local communities living alongside large carnivores in Africa's dryland landscapes. This model draws on our knowledge as scientists and practitioners of carnivore conservation: our understanding of coexistence varies based on our personal and disciplinary backgrounds, our field experiences, and the geographical contexts of our work (see also Kiik, 2018). Our Theory of Change is also informed by critical and place-based experiences of coexistence and well-being. After a brief introduction to the study context and description of the Theory of Change and its development, we detail the main pathways of change identified, their expected outcomes, and the assumptions on which they are based. We finish up with a discussion of potential frameworks through which the model may be locally implemented and operationalized. Throughout our analysis, unless otherwise stated, our use of the terms "costs" and "benefits" is intended to encompass the multiple dimensions of material, subjective, and social well-being, in line with Woodhouse et al.'s (2017) framework.

STUDY CONTEXT

Our analysis focuses on coexistence between people and large carnivores in Africa's drylands. These are defined as lands where annual precipitation is less than two thirds of potential evaporation, and range from subhumid areas through to hyper-arid deserts (Millennium Ecosystem Assessment, 2003). Africa's drylands extend across 43% of the continent's land mass (FAO,

2008) and are home to five species of conflict-causing large carnivores: lion (*Panthera leo*), leopard, cheetah, African wild dog (L*ycaon pictus*), and spotted hyena (*Crocuta crocuta*).

Our study context, therefore, stretches across a vast region, encompassing landscapes that are historically, politically, economically, culturally, and ecologically diverse and that are disproportionately impacted by climate change (IPCC, 2019). Local experiences of coexistence will vary considerably from place to place (Pooley, 2016) and across the range of species present within each area (Dickman et al., 2014). Our Theory of Change is therefore designed to provide a broad and flexible framework that can encompass different place-based contexts and facilitate management approaches that recognize and value a wide diversity of experiences of coexistence. It can be used in areas that still support populations of large carnivores, that provide corridors for such populations, or in areas of wildlife recovery.

METHODS

The Theory of Change Approach

A Theory of Change approach was chosen over other conservation decision frameworks (Bower et al., 2018; Núñez-Regueiro et al., 2020), as it is qualitative and relatively simple, yet can provide a big picture approach to help understand complex socio-ecological systems. Theories of change are process-oriented tools, they are particularly suited to development through expert and stakeholder consultation. In making explicit the logical connections and assumptions between activities, outcomes and impact, theories of change help facilitate an understanding of the pathways and steps through which interventions result in their desired impact, and allow testing of these assumptions (Biggs et al., 2017; Rice et al., 2020). They are widely adopted in the field of international development (Vogel, 2012; Valters, 2014) and are increasingly used in conservation to design, monitor and evaluate interventions (Biggs et al., 2017; Balfour et al., 2019; Rice et al., 2020; van Eeden et al., 2021).

Our Theory of Change was used to understand how a complex range of factors and their interactions can foster coexistence between people and large carnivores and to identify major pathways that can lead to change. It was generated through expert consultations, initiated in a workshop process, working backwards from the intended impact through to the changes, actions and conditions needed for its achievement. As a first step we developed a clear understanding of the issue at hand to identify the intended impact. In the second step we identified barriers to achieving the impact. For the third step we identified the various objectives or outcomes needed to overcome the barriers to deliver the impact, breaking down the changes that need to occur before the impact can be achieved. In the fourth step we listed the specific outputs, actions or interventions needed to bring about the identified outcomes. In the fifth and final step we reflected on and questioned the assumptions under which outputs and outcomes are believed to be linked. We structured outputs into overarching pathways and identified enabling conditions, or rather, principles and contextual elements, based around our assumptions, that determine the successful progression from an intervention through to its intended impact. Our approach follows that taken by Biggs et al. (2017) in their analysis of the illegal wildlife trade, and as such follows a uniquely tailored approach.

The Workshop

A workshop was designed with the express purpose of developing a Theory of Change to improve coexistence with large carnivores. The workshop took place at the Brackenhurst Conference Center, near Nairobi, in Kenya, in 2018 and was attended by 14 scientists and practitioners in carnivore conservation with experience covering the full guild of conflict-causing large carnivores (lion, leopard, spotted hyena, cheetah, and African wild dog) and across 19 countries in Africa. Two additional experts working in Europe and Asia also participated in the workshop to provide alternative perspectives and experiences from other regions. Participants included government, NGO and academic representatives, and most had substantial experience in multi-disciplinary research and/or practice. Three participants were leaders of community-based projects, and their lived experiences working within these communities helped inform the workshop. The framework for the Theory of Change was developed over 2 days, with the discussion facilitated and guided by SMD. The model was further developed after the workshop and simplified through remote discussions with workshop participants (**Figure 1**). Three additional coauthors participated in the writing process, bringing additional expertise, including in anthropology, monitoring, and evaluation. Details of contributors are provided in the **Supplementary Table S1**.

RESULTS

A Theory of Change to Enhance Coexistence Between Large Carnivores and Local Communities

In line with our understanding of coexistence as a dynamic state, but where interactions between people and large carnivores are kept within sustainable limits, we defined the desired impact of our Theory of Change as enhanced coexistence between large carnivores and local communities in Africa's dryland landscapes.

Outcomes

Coexistence is intrinsically a socio-ecological state, hence outcomes needed to enhance coexistence necessarily span both social and conservation goals (**Figure 1**). These outcomes recognize the multiple material, subjective, and social well-being components to coexistence (Woodhouse et al., 2017).

The achievement of sustainable large carnivore populations depends on a series of outcomes linked to material as well as other components of people's well-being. These well-being components concern the ability of local people to organize the use of natural resources, and to build stable, sustainable and resilient livelihoods to meet their needs and aspirations. In logical order, starting with lower level outcomes and working up to final outcomes, these include improved rule enforcement and compliance arising from agreed systems that underpin sustainable natural resource management (O.1), alongside measures that reduce

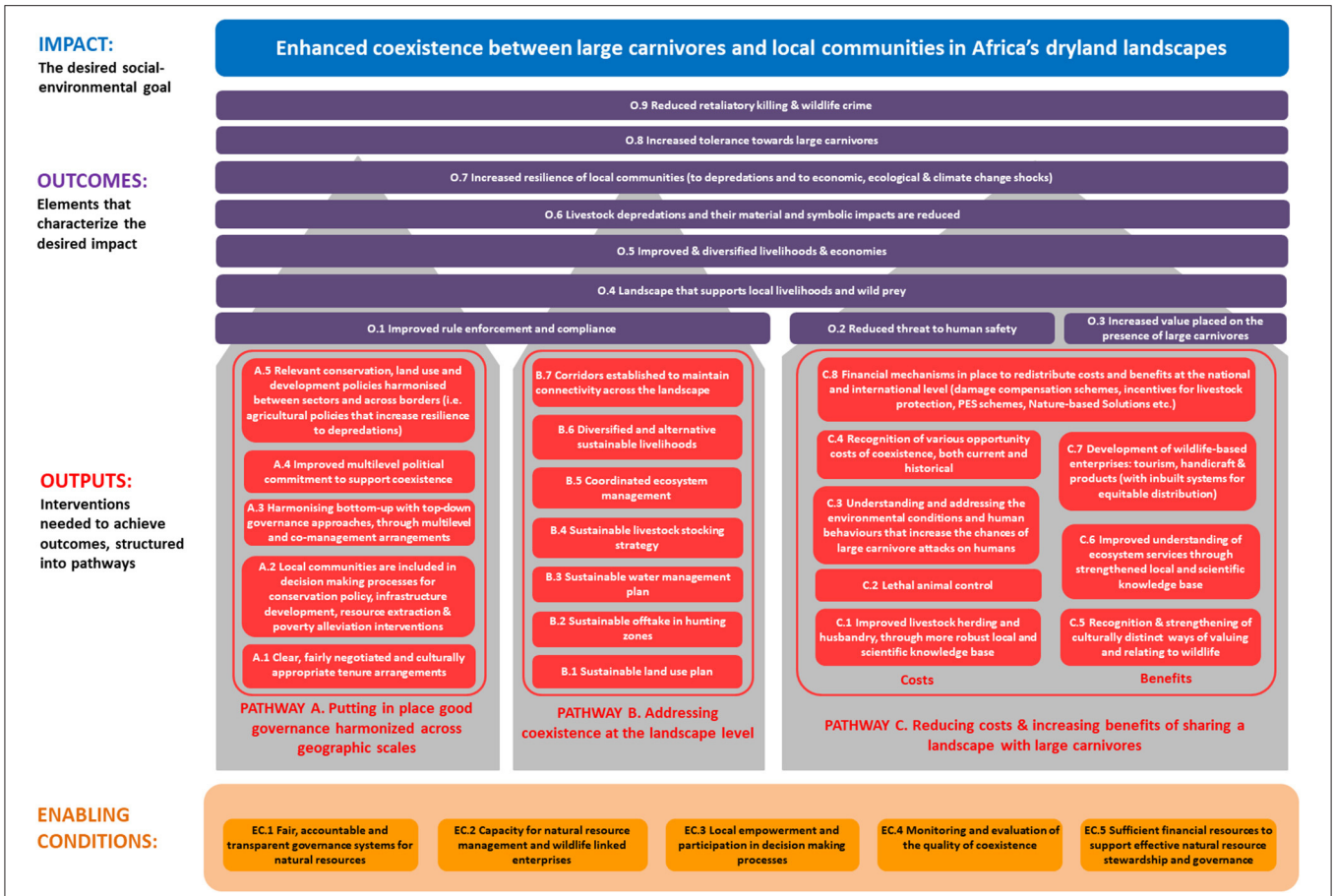


FIGURE 1 | The Theory of Change for enhanced coexistence between large carnivores and local communities in Africa's dryland landscapes.

the threats to human safety posed by large carnivores (O.2) while generating value for communities (O.3). Landscapes that support local livelihoods and wild prey (O.4) should result from sustainable rangeland management, while improved and diversified livelihoods and economies (O.5) should reduce reliance on natural resources. Communities bear costs of living alongside large carnivores, in addition to threats to human safety (O.2), and hence it is important that measures are taken to reduce to a minimum the material and symbolic impacts of these events (O.6). Climate change is projected to increase desertification and the frequency and intensity of extreme climate events, including droughts, dust storms and floods, all of which already exert catastrophic impacts on African dryland systems (Middleton and Sternberg, 2013; IPCC, 2019). Thus, the implementation of sustainable approaches to rangeland management should help to secure the resilience of communities in the face of climate change (O.7). Finally, these steps should result in increased tolerance toward large carnivores (O.8), and contribute to a reduction in retaliatory killing and wildlife crime (O.9). These outcomes together contribute to the overall impact of enhanced coexistence between large carnivores and local communities.

The outcomes we identify not only address material components of well-being, but also link to subjective components of well-being, as their delivery requires addressing issues of distribution, equity, and justice in environmental resource management and in conservation related policies. In addition, they link to social components of well-being because of the need to structure and mediate relations within communities and between communities and other actors, and to support livelihoods and practices tied to people's identity and to their sense of belonging and of place. Any changes in the material value of large carnivores will also, most likely, affect the subjective and social value of these species. These subjective and social dimensions of outcomes must be addressed, alongside material dimensions, in order to ensure that local interests and socio-ecological relations are recognized and valued. This requires a holistic consideration of "the complexity of people's lives, incentives and aspirations, which are both shaped by and shape their natural environment" (Woodhouse et al., 2017, p. 97).

The progression from pathways through to outcomes and impact relies on a series of assumptions that are difficult to examine in isolation, as they will interact with each other in different ways, depending on context. Therefore, rather than listing each separate assumption, we discuss them in the following section.

Pathways to Outcomes and Impact

Three pathways were developed that incorporate actions and interventions that lead to the expected outcomes. A set of enabling conditions are required to provide the underpinning foundations to support the pathways, and are key for effective, fair and transparent stewardship and governance of natural resources. They include systems for fair, accountable and transparent governance (EC.1); capacity (EC.2); local empowerment in decision making (EC.3); monitoring and evaluation (EC.4); and sustainable financial resources (EC.5). Supported by the enabling conditions, the pathways, taken together, are expected to enhance the socio-ecological sustainability of resource use, thereby increasing the resilience of livestock keepers to economic, ecological, and climate change shocks.

Pathway A—Putting in Place Good Governance Harmonized Across Geographic Scales

Pathway A aims to integrate local perspectives into different levels of decision-making by putting in place good governance and harmonizing interventions across governance scales. The pathway is intended to facilitate fair and equitable negotiation of rules over access and management of natural resources, including large carnivores, and also addresses larger processes that structure local economies with the aim of improving livelihoods. Developing governance approaches that can fairly and sustainably address the complexities of people's relations with nature has proven to be challenging (Roe et al., 2009), particularly given widespread government reluctance to devolve autonomy and control over natural resources to communities (Nelson et al., 2020). Self-governance and devolution are the guiding principles behind community-based natural resource management (Ostrom, 1990) and are considered essential for community-based conservation (CBC) arrangements to thrive (Nelson and Agrawal, 2008). However, under these principles, CBC success depends on communities valuing large carnivores as a resource, which is very often not the case. Moreover, even where carnivores are valued as a resource, the efficacy of CBC in delivering positive conservation outcomes for these wide-ranging species is limited by the local scale of CBC activities.

Governance approaches that involve higher levels of governance, through regional, national, multinational, and supranational engagement and through partnerships with NGOs, businesses or international networks (Lemos and Agrawal, 2009) are therefore necessary for several reasons. Firstly, they can accommodate the ecology and population dynamics of large carnivores, which extend beyond community boundaries, across national, and regional borders (Woodroffe and Ginsberg, 1998; Trouwborst, 2015; Durant et al., 2017). Secondly, they allow for national and international interests in conservation, and human and indigenous rights to be represented and negotiated, thus also facilitating the creation of multilevel partnerships and international community networks (Lemos and Agrawal, 2009; https://www.iccaconsortium.org). Thirdly, they can more adequately account for the global structural processes, both historical and ongoing, that underpin current patterns of poverty, inequality, and extractive resource use (Robbins, 2012; Moore, 2015). Global structures affect local communities and large carnivores in multiple ways, for example, driving the conversion of land with high value to biodiversity and pastoral communities for more intensive activities like agriculture or mining (Batterbury and Ndi, 2018). They can result in lasting changes to rural economies and agrarian practices, impacting the sustainability and resilience of community livelihoods and resource use strategies, for example by changing patterns of livestock predation and human-carnivore interactions (Lescureux and Linnell, 2013; Margulies and Karanth, 2018).

In order to reconcile these complex national, regional, and global processes with the place-based contexts relevant to people and their livelihoods, conservation of large carnivores requires governance that can empower and engage local communities in decision-making that extends beyond individual communities. Designing governance structures that span across different levels of scale is a challenge when the interests of local communities conflict with the interests of the national, regional or global community (Nelson et al., 2020). In such situations, systems of governance using a "freedom within frames" approach can be useful, whereby large carnivore management works within a nested hierarchy of governance structures (Linnell, 2005). Here, high-level policy frameworks provide general guidelines and principles and set boundaries within which lower levels can operate, while communities are able to make autonomous and locally adapted decisions within the limits of these frameworks (Linnell, 2005). Such systems need to account for legitimate grievances from those who shoulder the burden of living alongside large carnivores against the imposition of higher-level policy frameworks that limit their freedom, say, to manage or eliminate threats posed by these species (Linnell and Kaltenborn, 2019). High-level policy should also provide mechanisms that increase resilience of the socio-ecological system in response to climate shocks, that sustain coexistence, while supporting the adaptation of communities to a changing climate (Abrahms, 2021).

To be effective, governance frameworks need not only to be harmonized across geographic scales but should also provide mechanisms that empower communities to negotiate their interests at these different scales. Thus, the measures in this pathway address both bottom-up and top-down approaches to governing nature, and improve negotiations between them. Bottom-up approaches include local-scale governance of wildlife and natural resources through negotiation processes that integrate local knowledge into management (Folke et al., 2005; Armitage et al., 2009; Linnell, 2015; Redpath et al., 2017; Butler et al., 2019), and theoretical and practical facets of community-based conservation (CBC) (Nelson and Agrawal, 2008; Mishra et al., 2017). Progress in this pathway requires ensuring that local communities have meaningful roles in decision-making processes.

The interventions we identified within this pathway therefore aim to make coexistence approaches complementary and compatible across different levels and scales. They require clear, fairly negotiated and culturally appropriate tenure arrangements (A.1) and the inclusion of local communities in large-scale decision-making processes for conservation policy,

infrastructure development, resource extraction, and poverty alleviation interventions (A.2). This requires harmonizing bottom-up with top-down approaches as discussed above (A.3). These actions will help to build a multilevel political commitment to support coexistence (A.4), which requires local understandings of coexistence to be recognized, valued and integrated into large-scale planning. Finally, the relevant conservation, land use and development policies should be harmonized across geographic borders as well as between government sectors (environment, infrastructure, economic development, agriculture etc.; A.5). This includes, therefore, the development of transnational cooperation and, for example, of national and international agricultural policies that increase local resilience to livestock depredation (A.5). It also requires improving policy alignment at all levels, both within and between sectors, which should be based on a sound understanding of the interactions between micro and macro level structures that govern processes of resource extraction and conservation (Agrawal and Ostrom, 2006; Igoe, 2006), as well as adaptation to climate change (IPCC, 2019). Adaptable multilevel solutions are needed that can integrate complexity and facilitate the dialogue, information exchange, cooperation, and negotiation needed to establish both upward and downward accountability (Cash et al., 2006; Berkes, 2009; Butler et al., 2019, 2021). The success of these interventions depends on cooperation between neighboring land management authorities and owners including, where appropriate, protected area management authorities.

Adaptive co-management has been proposed as an approach to governance systems that can navigate across different geographic scales while addressing the inherent complexity of socio-ecological systems and their associated uncertainties (Folke et al., 2005; Armitage et al., 2009; Butler et al., 2019, 2021). Adaptive co-management has been defined as "a process by which institutional arrangements and ecological knowledge are tested and revised in a dynamic, on-going, self-organized process of trial and error" (Folke et al., 2002, pg. 20; Plummer and Armitage, 2007; Berkes, 2009; Plummer and Baird, 2013; but see also Butler et al., 2019). By linking actors horizontally and vertically (Plummer and Baird, 2013), as is essential for successful large carnivore management, adaptive co-management supports communities to become managers of natural resources, to invest in long-term sustainable management of ecosystem services, and to make informed and difficult trade-offs to support their long-term well-being (Fabricius et al., 2007). Adaptive co-management is also implicitly dynamic which, crucially, enables it to be agile and flexible in the face of climate change.

Pathway A, therefore, fundamentally, aims to tie together multiple approaches to governance in order to secure community engagement and ownership in decision-making, and to ensure that stewardship can be coordinated at the large geographic scales needed for the survival of large carnivores.

Pathway B—Addressing Coexistence at the Landscape Level

Interventions within pathway B aim to improve the governance and stewardship of natural resources at the landscape level. This pathway emphasizes building and strengthening local institutions at scale in order to improve enforcement and compliance with rules about the use, management, and conservation of resources important to both local livelihoods and large carnivore habitat and prey requirements (Agrawal and Gibson, 1999). This includes a series of interventions targeted at sustainable rangeland management that are based on an understanding of coexistence between people and large carnivores embedded within a broader set of socio-ecological relations (Ghosal et al., 2015; **Figure 1**). A focus on the landscape level enables conservation interventions to be directed not just toward carnivores themselves but also toward their habitat, wild prey and interspecific interactions. Landscape approaches to conservation also enable the integration of cultural, political and ecological considerations so that humans, their livelihood practices, and everyday tasks, are understood as integral elements of the local ecology (Sayer et al., 2013).

The interventions we identified in this pathway involve developing community-led natural resource management plans that are both ecologically and socially sustainable. These include plans for ecosystem management (B.5), including livestock stocking strategies (B.4), water extraction (B.3), hunting (B.2), and different types of land use which may include protected areas (B.1). Underpinning these management plans is the need for clear, fairly negotiated and culturally appropriate tenure arrangements addressed under pathway A (A.1; Western et al., 2020). Tenure arrangements may concern individual or communal ownership, control, access, use of land, and natural resources, including rights to include and exclude outsiders from key resources such as hunting grounds, grazing land and water, or from tourism development (Ostrom, 1990; Bluwstein, 2017; Homewood, 2017). In some cases alternative and diversified sustainable livelihoods (B.6) may also play a role in reducing pressure on natural resources (Roe et al., 2015). Finally, interventions to safeguard wildlife movement corridors to maintain connectivity across multiple-use landscapes (B.7), including careful consideration of any fencing interventions (Durant et al., 2015), will be needed to secure the viability of large carnivore populations and increase their resilience to climate change. This requires collaboration between adjacent communities and regions, and with protected area managers and policy-makers, to combine and harmonize community-level management at the landscape level, integrated with recent information on environmental change. These interventions together involve the engagement of institutions that structure relationships between local actors, and relationships between local communities and external actors, such as other communities, NGOs, private companies and states (Pathway A; Agrawal and Gibson, 1999). Approaches could build on existing community-based natural resource management or similar frameworks adopted across Africa's semi-arid landscapes (Nelson et al., 2020).

The importance of recognizing the diversity of norms and interests within and between communities is particularly important for this pathway, as groups, subgroups, and individuals may have different priorities for resource use and distribution. Negotiations over resource access will also have impacts across different dimensions of well-being, with

implications for the material well-being of communities and individuals, for their perceptions of equity and justice, and for their ability to maintain culturally specific socio-ecological relations that encompass traditional livelihoods and practices (Martin et al., 2016; Oldekop et al., 2016; Lichtenfeld et al., 2019). Local notions of identity and stewardship are often centered around natural resource use, and underpin cultural dimensions of well-being, hence an understanding of the cultural diversity of perceptions relating to conservation and sustainability needs to be integrated into management (Lewis, 2008; Homewood, 2017). This will require an understanding and recognition of current and historical grievances, which may include exclusionary practices such as the gazetting of protected areas (Brockington and Igoe, 2006; Büscher and Fletcher, 2020). For example, new forms of land grabbing by multiple powerful actors, including the state and multinational corporations, primarily for large-scale agricultural production (Dell'Angelo et al., 2017) but also for conservation purposes, risk reproducing past injustices and further alienating local communities (Homewood, 2017; Davis et al., 2020).

Pathway C—Reducing Costs and Increasing Benefits of Sharing a Landscape With Large Carnivores

Interventions within pathway C aim to reduce the costs and increase the benefits of coexistence by raising the economic and cultural value placed on the presence of large carnivores; reducing the threat of large carnivores to human welfare and safety; and providing linked support and funding streams to enable local communities to improve or diversify their livelihood strategies (**Figure 1**). The pathway also aims to redress the inequitable distribution of costs and benefits at local, national and international scales. This depends on understanding what an equitable distribution might mean to a community and to its different members. Moreover, it relies on the multilevel structures discussed in pathway A, through which distribution can be fairly and equitably negotiated, alongside securing sustainable stewardship at a landscape level through pathway B.

A key technical intervention in this pathway is aimed at improving livestock herding and husbandry practices to reduce livestock injury or loss (C.1). Livestock depredations and the implementation of measures to prevent them can result in significant financial, labor and emotional strain for farmers (Inskip and Zimmermann, 2009). Measures to prevent attacks on livestock should be co-designed with livestock keepers, prioritizing herder knowledge and experience, while providing scientific, technical and material support to build on existing capacity (Lichtenfeld et al., 2019). This enables interventions to be adapted to on-the-ground physical, economic, labor and cultural opportunities, and constraints. Another intervention provides for the possibility of legal avenues for targeted lethal control (C.2). Carefully managed lethal control may be an important management tool where local socio-ecological relations rely on notions of reciprocity and control or on longstanding hunting traditions (Goldman et al., 2010; Lescureux et al., 2011). Moreover, it may be necessary under conditions of intense conflict or imminent threat to human safety, where other solutions have proved futile or are not available (Packer et al., 2019). The removal of individuals that are causing problems to local communities will necessarily have to be balanced against the conservation status of the species or population in question, and exceptional circumstances may require negotiation between trade-offs. Understanding the environmental conditions and human behaviors that increase the likelihood of large carnivore attacks on humans (C.3) is also crucial for the development of new approaches to their prevention, and for mitigating interactions with carnivores that have the potential to be highly traumatic for local communities.

Interventions aimed at addressing the costs of carnivore conservation borne by local communities must also recognize that these often extend beyond the economic impact of livestock depredations. Historical and contemporary opportunity costs (C.4) may include foregone revenue from land use change, more intensive livestock stocking strategies, or hunting. For many, the collective memory and ongoing trauma of land dispossession and community displacement, through the creation of national protected areas or private hunting or tourism reserves, may be the dominant lens through which carnivore conservation interventions are perceived (Neumann, 2001; Brockington and Igoe, 2006; West et al., 2006; Homewood, 2017). Carnivore conservation strategies aimed at engaging local communities must consider the legacy of such past conservation projects, including their links to colonial history and state-building endeavors. Top-down approaches may reconfigure the local political landscape in unexpected ways, and reinforce the perception of carnivores as symbols of state, foreign or elite power (Duffy et al., 2019). Militarized approaches to conservation, in response to the global demand for wildlife products, may also contribute to community alienation. In the long-term, community-based and supported policing of illegal activities, including the incorporation of value-based approaches to anti-poaching, may secure better protection and help prevent the escalation of violence (Neumann, 2004; Duffy et al., 2019). Carefully designed outreach and awareness raising campaigns can also decrease illegal activities (Holmes, 2003; Steinmetz et al., 2014; Biggs et al., 2017; Chen et al., 2019), and underpin good governance and effective stewardship of natural resources to enhance coexistence.

Alongside the costs in pathway C, are a series of interventions that aim to increase the benefits of coexistence. The first two interventions are based on recognizing and strengthening culturally distinct ways of valuing, benefiting from and relating to wildlife (C.5). As mentioned previously, perceptions and attitudes toward carnivores are diverse, multi-layered, and often ambivalent (Goldman et al., 2010; Baynes-Rock, 2013). In practice, incorporating local community needs into large carnivore conservation may mean integrating scientific discourse on ecosystem services with local understandings of nature and of sustainability (C.6). These may include the provision of secure livelihoods, balanced herbivore populations, access to food, safe water, mineral licks, and grazing land, and extending further to notions of social resilience, spirituality, identity, stewardship, socio-ecological diversity, community, and sovereignty (Diaz et al., 2018). Ethnographic studies have explored people's sensory, spiritual and emotional connections with wildlife, demonstrating

the complex ways though which local people care for and benefit from their environment that extend beyond the realm of economic profit (Singh, 2018). Greater attention should be paid to these culturally rooted socio-ecological relations, as these represent place-specific forms of coexistence and conservation in their own right (Sandbrook, 2015), and may further be harnessed as foundations to forge synergistic partnerships with external conservation organizations (Peterson et al., 2010). However, there are perils in cherry-picking congenial coexistence narratives and practices, without recognizing their diversity on the ground, their evolving nature and how they are shaped by politics. Culturally-rooted relations with nature may be co-opted in local power struggles (Brockington, 2006), representing what Homewood (2010, p. 179) refers to as "politically loaded statements about identities and aspirations." This highlights the importance of engaging with local perceptions of wildlife on people's own terms, acknowledging the existence of a full range of beliefs and practices that may be at times beneficial and at other times harmful to conservation (Homewood, 2010).

Wildlife-based enterprises provide mechanisms to generate material, cultural, and social benefits from the presence of wildlife (C.7). Wildlife tourism is the most widely used intervention to enable local communities to secure benefits from sharing a landscape with large carnivores. Notable examples of tourism for carnivores and other charismatic species across Africa have been promoted as conservation and development successes (Lindsey et al., 2013). Whilst most offer wildlife photography-based safaris, some also allow trophy hunting or include handicrafts or other wildlife-based products (Mishra et al., 2003; Lindsey et al., 2006). However, individual schemes may function through very different levels of engagement with communities, the state, NGOs and the private sector. Positive experiences, where wildlife tourism contributes to combined social, ecological and economic outcomes, have been recorded, particularly for well-designed, long-term projects (Brooks, 2017). For example, in Namibia, long-term CBC schemes in conservancies on marginal land have benefitted from technical and financial contributions from external agencies, and have provided more tangible benefits to local people than alternatives (Dressler et al., 2010). However, some tourism-based CBC schemes have faced criticism regarding their ability to provide benefits that are sufficient to outweigh the costs of living alongside problematic wildlife, and to reach those who shoulder the greatest burden of coexistence (Songorwa, 1999; Gandiwa et al., 2013).

Many places lack the well-developed infrastructure and political stability required to attract and accommodate tourism, which makes development through wildlife tourism unrealistic (Walpole and Thouless, 2005; Brito et al., 2018). The recent Covid-19 pandemic has also had drastic impacts on tourism revenue generation across Africa, and highlights the volatility of the tourism industry and the risks associated with an over-reliance on tourism to deliver conservation and community development (Lindsey et al., 2020). Even where tourism thrives, tourism experiences targeted at foreign visitors may provide a depiction of wilderness that is disconnected from its historical and social context, that conflicts with local conceptions of nature, and exacerbates nature-society divisions in the context of global uneven development (Büscher and Fletcher, 2020). The cultivation of more engaged, long-term, local, and every-day nature-based experiences is likely to be critical to ensuring a more reliable and sustainable tourism sector and to democratize access to nature (Vannelli et al., 2019; Büscher and Fletcher, 2020; Lindsey et al., 2020). The promotion of environmental education at community and national levels, aimed at valuing biodiversity, traditional knowledge and existing biocultural relationships may provide cultural and conservation benefits that extend beyond economic profit, highlighting also the importance of creating opportunities for nature-based experiences that are accessible to local and domestic residents (Black, 2016; Büscher and Fletcher, 2020).

A series of financial mechanisms exist to redistribute the material costs of coexistence on a national and international level (C.8), including incentives for livestock protection (van Eeden et al., 2018) and damage compensation and insurance schemes (Dickman et al., 2011). However, the success of damage compensation schemes in increasing tolerance for damages appears to be limited, particularly when they are applied in isolation (Agarwala et al., 2010). Nonetheless, such mechanisms may help to demonstrate a wider political commitment toward sharing the costs of carnivore conservation (Agarwala et al., 2010; Dickman et al., 2011). Importantly, they should be incorporated into a holistic approach to large carnivore conservation that avoids unintended outcomes, as they otherwise risk removing incentives to safeguard livestock against attacks or disrupting important cultural values associated with large carnivore conservation (Nyhus et al., 2003).

Opportunities to support more integrated approaches to nature stewardship may be available through broader schemes that provide financial benefits from sustainable stewardship of natural resources, including payments for ecosystem services (PES), such as through carbon-based schemes (African Development Bank Group, 2015; Kiffner et al., 2019), and proposals of universal basic income (Fletcher and Büscher, 2020). The economic principles of these financial mechanisms can vary from market- to welfare-based arrangements, depending on the funding sources and how they are harnessed and distributed within communities. Debates regarding the efficacy of market-based PES schemes in delivering global and local ecological and social benefits are heated and ongoing (Corbera, 2012; Fletcher et al., 2016; Fletcher and Büscher, 2017; Ferraro, 2018). Many of the pitfalls encountered by PES schemes at the community level mirror those encountered by CBCs (Roe et al., 2009), centering around issues of governance, transparency, tenure, equitable benefit distribution, and ability to secure long-term funding (Dougill et al., 2012; Corbera et al., 2019). However, opportunities for generating income from sustainable stewardship are likely to become more widely available in future. Nature-based solutions (NbS) have recently attracted considerable international attention, and could provide substantial financial resources for local communities to support responsible nature stewardship (Pettorelli et al., 2021; Wood et al., 2021). Moreover, debates over climate and ecological justice have seen growing calls for an explicit recognition of the ecological debt accumulated by the global north through centuries of colonial domination and

resource exploitation in the global south (Bellamy Foster and Clark, 2004). Proponents of ecological and climate reparation have brought forth solutions that include the restructuring or canceling of financial debt owed by the global south, and the creation of a Global Climate Stabilization Fund and Resilience Fund Programme, dedicated to meeting needs under the loss and damage category of financial compensation, and funded by the countries most responsible for global ecological and climatic destabilization (Perry, 2020). As concerns for climate change galvanize opportunities for financial redistribution and reparation, it is important that emerging institutional and funding arrangements are designed to address the biodiversity crisis alongside the climate crisis, while also contributing to local communities' well-being and resilience (Seddon et al., 2020; Pettorelli et al., 2021). The development of holistic approaches to address these joint social and environmental challenges will require global commitment (Pettorelli et al., 2021; Seddon et al., 2021), and a careful attention to how these approaches are implemented on the ground (Dougill et al., 2012).

DISCUSSION

Our Theory of Change identifies multiple social and ecological goals, or outputs, leading to enhanced coexistence between large carnivores and local communities, and three broad pathways through which they can be achieved, namely: (A) putting in place good governance harmonized across geographic scales; (B) addressing coexistence at the landscape level; and (C) reducing costs and increasing benefits of sharing a landscape with large carnivores. The model is based on our knowledge as conservation practitioners and scientists, including our everyday experience of conservation challenges on the ground, and draws on the literature on conservation and coexistence. Our intent is to develop a framework as broad and comprehensive as possible, able to identify linkages between the overall intended impact of enhanced coexistence with specific outcomes and their pathways, and to report on key debates concerning these links. However, coexistence is by definition a situated experience, embedded in a place-based socio-ecological context, but which will also be subject to change, including changes due to a warming planet. Therefore, to be relevant, our overarching Theory of Change has been designed to be flexible, to enable local adaptation through meaningful community engagement with shifting place-based realities as they are experienced by communities and large carnivore populations.

The complexities inherent in people's relationships with nature, and particularly with large carnivores, mean that our Theory of Change is unlikely to reach an end point whereby sustainable coexistence is achieved. Rather, coexistence requires an ongoing process of negotiation that recognizes the diverse and changing relationships between large carnivores and local people, and identifies interventions that can minimize costs and increase benefits in ways that foster tolerance (Linnell, 2013). Climate change provides an additional layer of complexity, further modifying relationships between people and large carnivores in ways that may be difficult to predict (Abrahms, 2021). Our model thus represents the actions, structures, and processes that should be put in place to allow this negotiation to take place, and which move the tolerance "dial" toward enhanced coexistence, while allowing for change. This includes conservation and natural resource management governance structures that incorporate local interests, support sustainable management of habitat and wildlife, and ensure equitable distribution of the costs and benefits of living alongside large carnivores.

Well-being-based approaches offer a useful framework for involving local communities in Theory of Change adaptation processes and defining, qualifying, and monitoring implementation (Woodhouse et al., 2017). Woodhouse at al. (2017) provide examples of how communities have been engaged through participatory methods to define their own material, subjective, and social needs and aspirations, in order to identify relevant and meaningful social and ecological goals. Including people from different age, gender, class, ethnicity, and livelihood groups in the adaptation process helps address heterogeneity in local values and needs (de Lange et al., 2016). It also helps identify vulnerable groups who are most adversely affected by large carnivores to ensure that they are the main beneficiaries of designed interventions (Woodhouse et al., 2017). The goals identified by local communities can be incorporated into the desired outcomes of our Theory of Change, and used, and adapted as necessary, as part of a cycle of monitoring, evaluation, learning and adaptation (Lichtenfeld et al., 2019).

Large Carnivores, Communities, and the Problem of Scale

Pathways B and C represent a two-pronged approach to address site-based actions needed to enhance coexistence with large carnivores through the sustainable management of drylands and mitigation of conflict, while pathway A provides the governance pathway on which their success depends. Crucial to this pathway is the reconciliation of locally based governance with approaches that are able to work at the large geographic scales needed for large carnivore conservation. This points to the need for a "freedom within frames" approach, that supports meaningful community engagement with a wide range of stakeholders to achieve solutions that deliver social and ecological benefits across multiple scales.

Beginning in the 1980s, CBC approaches have provided mechanisms of governance at the local level. CBC is based on an ethos of participatory engagement through the inclusion of traditional knowledge and community interests in resource management and has been implemented across the African continent (Dressler et al., 2010). Key to CBC is the generation of economic benefits to local communities from nature protection and the provision of ecological services that provide income, socio-economic development and poverty alleviation (Büscher and Fletcher, 2020). CBC approaches, however, have often fallen short of delivering the social and ecological benefits initially claimed (Agrawal and Gibson, 1999; Songorwa, 1999; Newmark and Hough, 2000; Adams et al., 2004; du Toit et al., 2004; Galvin et al., 2018). They have been critiqued in the literature for their tendency to ignore the heterogenous

nature of communities and the power dynamics within them, obscuring the presence of multiple actors and interests within a community as well as the existence of both winners and losers of conservation and development interventions (Agrawal and Gibson, 1999). Several examples exist of cases where the domination of powerful state, private and elite interests further undermine already weak incentives, so that the benefits are often not sufficient to outweigh the opportunity costs of conservation for most community members (Dressler et al., 2010; Bluwstein, 2017; Homewood, 2017). Moreover, anthropological and ecological research has countered simplistic depictions of local and indigenous communities as natural stewards of their environment and its wildlife (Heatherington, 2010). This is further complicated by the fact that local communities, their traditions and their practices change over time, often in response to larger structural changes. For example, in many cases, colonial and state expansion and wider transitions into capitalism, have disrupted traditionally low impact and largely sustainable natural resource use and livelihoods (Robbins, 2012; Brightman and Lewis, 2017).

Perhaps the main critique to CBC is the limited extent to which it links to larger scale governance approaches, which limits meaningful control of communities over natural resources, due to conflicts with the interests of other stakeholders, including governments and private investors (Hutton et al., 2005). In this context, it is important that carnivore-focused conservation NGOs consider interventions that build trust and support from local communities even if they only provide marginal indirect benefits to large carnivores, since such interventions may ultimately enable broader and more direct outcomes (Young et al., 2021). This requires understanding and respecting local priorities even when they appear to depart from carnivore conservation goals. It also requires a long-term engagement. For example, technical and financial support from African People & Wildlife provided to small community projects, including water troughs, invasive species removal, and pasture beaconing and demarcation, demonstrated a genuine interest in community priorities. Such measures build positive relationships and trust with community decision-makers, and can be key to laying the groundwork for constructive discussions around carnivore conservation and coexistence (Mishra, 2016). Putting in place these building blocks of trust, shared goals, and mutual respect is critical for the effectiveness of interventions (Young et al., 2021), and can strengthen enabling conditions identified in our model, by supporting local governance, capacity and empowerment. In the challenging context of human-large carnivore coexistence, such approaches also support the establishment of strong democratic institutions that govern access and sustainable use of natural resources, which are founded on stakeholder dialogue and negotiation and facilitate compromises and synergies between local interests and conservation (Homewood, 2010).

Adaptive Co-management

As we have seen, large carnivore management is a dynamic and conflict-ridden space, where large carnivores can have serious impacts on local communities, and may even threaten public safety (Packer et al., 2019). The successful implementation of our Theory of Change will depend on management and governance frameworks that are able to mitigate conflicts between top-down large-scale policy and bottom-up local CBC-type approaches. This requires the empowerment and incentivization of local communities, alongside fair and equitable delivery of wider public goods such as biodiversity conservation (Brooks et al., 2013). The Theory of Change model provides a useful framework for an adaptive co-management approach, whereby the model can be used to structure on-going iterative learning integrated within participatory management of natural resource systems. This includes updating the model to address underlying uncertainties as more data become available, allowing adaptation and improvement in the model over time. In such an approach, knowledge and power are shared between stakeholders and conflict resolution is addressed dynamically through co-managed processes.

Approaches that may be helpful in initiating adaptive co-management structures for large carnivore conservation are procedures to obtain free and prior informed consent (FPIC) (Lewis et al., 2010; Buppert and McKeehan, 2013) or the social-license to operate (SLO), originally designed to negotiate over interventions that may not initially be welcomed by all members of a community (Kendal and Ford, 2018; Butler et al., 2021). Such frameworks have been recommended, for example, as an approach to rewilding by facilitating community engagement and negotiations between local interests and rewilding initiatives that may include restoration of large carnivores (Butler et al., 2019). An SLO could be used, for example, to negotiate community agreement on acceptable limits for the large carnivore population, in order to secure social acceptance.

Butler et al. (2019) propose an adaptive co-management cycle that uses participatory community and stakeholder engagement to develop agreed socio-ecological desired outputs (FPIC or SLO), and puts in place a series of interventions to achieve these outputs and a monitoring and evaluation plan to measure progress, allowing ongoing adaptation in line with learning. Crucial to the successful operation of this governance system is the establishment and maintenance of a facilitation team regarded as independent and trustworthy by all stakeholders (Cooke and Kothari, 2001). The facilitation team is responsible for identifying and engaging stakeholders, enabling dialogue and consensus-building, brokering knowledge and information and mediating any conflicts that may arise (Pound, 2015; Butler et al., 2019). The effectiveness of such an adaptive management system depends on an ongoing review of the Theory of Change to ensure that assumptions behind interventions are valid and that gaps are identified and addressed (Lichtenfeld et al., 2019). This governance system would then facilitate interactions between communities and larger scale boundary setting and can incorporate multidimensional goal setting in ways that can be adapted and adjusted over time, including in response to a changing climate (Todd, 2002; Mishra et al., 2017; Butler et al., 2021).

The development of improved governance systems such as those described above, that can support sustainable approaches to coexistence with large carnivores across the scales needed, are crucial for success. Such approaches need to equitably

reconcile different and legitimate sets of interests that may conflict within and between local communities and with broader national and international public interests. At the same time, they need to avoid the widespread problems that typically contribute to governance failure, such as corruption and elite capture (Linnell, 2015). This will require financial and technical support to strengthen capacity for governance and provide incentive structures to foster sustainable natural resource use. It is important, however, to avoid perfection being an enemy of the good. Whilst our aim was to design a model that was ambitious and comprehensive, our Theory of Change is meant to be adaptable to local financial, political, and other contextual constraints. It can be used to prioritize key interventions that are effective, practical and realistic in any given context, and additional interventions can be incrementally introduced during adaptive co-management cycles, as needed.

CONCLUSIONS

Our Theory of Change is intended to provide a broad framework that can be adapted to the specifics of local contexts. It can also be coordinated between communities to provide a harmonized framework at the wide geographic scales needed for large carnivore conservation. The Theory of Change will require ongoing monitoring and evaluation within an implementation framework, such as within an adaptive co-management approach, to test assumptions and address underlying uncertainties, while also responding to climate change. Further research is also needed to improve the underlying knowledge that forms the basis of the model and, here, reporting project failure should be regarded as just as valuable as reporting project success (Catalano et al., 2019). Our model has focused on large carnivores in African drylands, however, the Theory of Change can be readily adapted to other taxa and other systems.

Ultimately, the experience of sharing a landscape with carnivores is likely to improve only when communities are allowed to influence the terms and conditions of coexistence. This requires creating and strengthening institutions through which local people can discuss and prioritize management interventions, exploring incentives, facilitating systems of local rule enforcement, and centering local people within adaptive co-management roles (Homewood, 2017). CBCs provide a useful starting framework, but need improvement to secure genuine community self-determination and to allow their evolution into transparent and accountable approaches able to deliver social equity and justice alongside ecological outcomes (Homewood, 2017). Corruption, elite capture, privatization, rent seeking and resistance to decentralization by governments may also need to be addressed at local and landscape scales in order to secure meaningful community sovereignty (Nelson and Agrawal, 2008).

Even more challenging are contexts where incentives for carnivore conservation are very weak and conservation ranks low in local and national priorities. Our modeled pathways to enhanced coexistence recognize that conservation success relies on a holistic approach that incorporates avenues through which local well-being and priorities may be recognized, understood and valued in tandem with conservation goals. Tackling coexistence at the landscape level requires acknowledging that relations between carnivores and people form part of wider negotiations over land and resource use, and that people's livelihoods and aspirations should be viewed as integral elements of a socio-ecological system. Sustainable coexistence will depend on the harmonization of bottom-up community-led approaches, with top-down regulation that allows conservation to be effective at scale, but provides sufficient autonomy to be acceptable to communities.

Critical to success for our Theory of Change is for wider global structures to fully recognize the costs of coexistence born by local communities, and find new and sustainable financial mechanisms to ensure that the global value attached to large carnivores is transferred to those communities that pay the costs of living alongside them. This requires rethinking current economic arrangements, such as addressing issues of climate and ecological justice, as well as valuing culturally rooted relations with nature. Large carnivores, because of their potential to act as umbrella or flagship species (Belbachir et al., 2015), can help to secure international public support for new global financial mechanisms that translate the global value attached to large carnivores into local benefits (Rands et al., 2010; Durant et al., 2017). The challenge that climate change poses to communities across Africa's rangelands, could then be used to allow communities to harness financial mechanisms that support nature-based solutions to the ecological and climate crises. Such mechanisms will need to be carefully tied to local community concerns, which may often be linked to non-monetary values, to ensure successful community conservation (Davis and Goldman, 2019). Trust in the approach will depend on clear and transparent dialogue, including acknowledgment of situations when local needs and priorities do not align with conservation goals. Overall, we hope this framework will help drive transformative change in the implementation of holistic approaches to conservation and development that are grounded in trusting and stable partnerships with local communities. The global response to climate change presents an impetus to initiate such change, that includes rethinking our relationship with nature, and providing tangible value that supports sustainable coexistence for communities who are directly dependent on natural resources.

DATA AVAILABILITY STATEMENT

The research contains a meta-analysis of existing, public research. Further inquiries can be made to the corresponding author/s.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants in accordance with the local legislation

and institutional requirements. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements. Ethical review and approval was not required for the animal study because the research provides a meta-analysis of existing research, leading to the development of a Theory of Change model.

AUTHOR CONTRIBUTIONS

SDu was responsible for the conception and design of the study, facilitated the workshop to develop the Theory of Change. SDu, JL, AO-C, SDI, SDo, PF, LH, JH, DI, AI, BK, LL, JM, NM, AS, and GY participated in the workshop and contributed to the design of the Theory of Change and its assumptions and theoretical underpinnings. AM synthesized and simplified the Theory of Change, identified the underpinning published literature. AM provided the first draft and SDu wrote the discussion. JL provided key concepts and ideas that informed the writing of the manuscript. All authors contributed to manuscript revisions, with the overall coordination of editing handled by SDu and AM.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2021.698631/full#supplementary-material

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Coexistence in Times of Climate Crisis: A Participatory Mapping to Understanding Conservation Conflicts in the Central Andes of Chile

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Purpose: Human-wildlife conflicts worldwide are arising, representing significant challenges for conservation biologists, decision-makers, and agropastoralist communities. Extreme climatic events, disease outbreaks, and land-use change could be intensifying these conflicts. The multi-species and mountainous landscapes seem prone to conflicts due to a lack of territory planning. These complex, dynamic, and multi-layered conflicts require a multidimensional approach. Currently, in Chile, mountainous landscapes have several threats, such as a lack of territory planning, mega-mining projects, and recently (last 10 years) the effects of the Mega-drought. Many transhumant agropastoralists have been forced to quit their livelihoods while increasing livestock-wildlife conflicts. We aim to build territorial planning within a holistic approach to strengthening the agropastoralists' competence to coexist with local wildlife (puma, condor, and guanaco) in Central Chile's Andes mountains.

Methodology: We conducted participatory mapping workshops with two agropastoralist communities in 2020. They were randomly divided into 4 to 7 people groups and told to draw a map representing their territory, including four elements: (1) natural and human components of the landscape, (2) natural wildlife conflict areas, (3) active grazing areas, and (4) their ideal future scenario, regarding their activity.

Findings: Results showed different spatial perceptions of the natural and human components of the territory. All agropastoralists (100%) indicated similar wildlife conflict areas: focusing in the Summerlands. All agreed that Mega-drought was the primary threat to their production, increasing the conflict with wildlife. Summerland areas are identified as suitable areas for working in conflict with wildlife.

Research Limitations/Implications: This study highlights the need for a multidimensional approach to conflict and territory planning to address conservation conflicts. The study's implications show that agropastoralists decided to reduce Summerland use and improve Winterland planning to increase livestock productivity

and reduce conflict with wildlife. Participatory mapping could help to prioritize areas to mitigate conflicts with wildlife.

Originality/Value: This study is the first in the Aconcagua valley to conduct a transdisciplinary & participatory approach toward coexistence between transhumant agropastoralists and wildlife. It also provides a baseline for similar schemes in semi-arid and mountainous landscapes worldwide facing rapid climate shifts and increasing human-wildlife conflict.

Keywords: Mega-drought, conservation conflict, perceptions, territory planning, coexistence

INTRODUCTION

Human-wildlife conflicts are arising worldwide (Marchini and Crawshaw, 2015; Badola et al., 2021; Naha et al., 2021). These conflicts could emerge when the presence or behavior of wildlife is an actual (or perceived as) threat to human interests or needs. Leading, thus, to negative impacts on people and/or wildlife. Human-wildlife conflicts could also be seen as an interaction between humans and wildlife that negatively affects an ecosystem's elements (IUCN, 2020a; Shanko et al., 2021). The origins of conflict situations are dynamic and manifold: humans expanding into protected areas and wildlife expanding into human areas (König et al., 2020, 2021). These conflicts involve various species, often adversely affecting communities. They pose severe challenges to governments and organizations to balance wildlife conservation and sustainable development (IUCN, 2020a). Moreover, these conflicts often originate from discordant interaction between wildlife and human activities (livestock, agriculture, others) (Redpath et al., 2013). Nonetheless, human-wildlife conflicts demand to be considered through the conservation conflict concept. Conservation conflicts often define a clash of two parties' interests (regarding conservation decisions). It seems problematic to think of wildlife as conscious antagonists in a conflict (Peterson et al., 2010). So, it is imperative to address the stakeholder's vision from different sides in emerging conservation conflicts and to provide evidence to face the diverse facets of a human-wildlife (and conservation) conflict and coexistence (König et al., 2020).

Nonetheless, every human-wildlife (and conservation) conflict is unique. Even if the settings appear similar, their characteristics require different approaches to solving them (Zimmermann et al., 2021). Conservation conflicts involving multi-species and complex scenarios such as mountainous areas and extensive territories are the most significant (Karanth et al., 2012; Pozo et al., 2020) and will require novel approaches and efforts from diverse stakeholders to deal with said complexity. A participatory approach would provide that novelty (Senbeto Jiren et al., 2021) by sharing the local knowledge and the scientific advances, considering the specific context (e.g., ecosystem type, involved species, underlying socio-political aspects, shifting climate, disease outbreaks, and land-use change). This is to co-produce feasible and adoptable mitigation initiatives (Hodgson et al., 2015; IUCN, 2020b; Sahraoui et al., 2021).

On the contrary, a single-species approach to a multi-species conflict is unlikely to reduce negative attitudes toward wildlife (Suryawanshi et al., 2013), and a lack of a participatory approach could lead to poorly understood reality, generating interest loss, distrust from the community, and nonconsensual solutions (Palomo et al., 2011), in this case, for conservation conflicts. Some authors from different geographic locations noted that participatory approaches are helpful tools to generate collective local information to orient further mitigation initiatives regarding a conflict. Still, these approaches often face the challenge of involving a sufficient amount and type of participants (Marino et al., 2021). A study in Italy regarding the interaction between livestock and carnivores stated that participatory approaches better acknowledge different actors' objectives. However, they urge to consider as many actors as possible to avoid a problem reduction (Marino et al., 2021). In another case in the Kibale National Park in Uganda, authorities built physical actions of mitigation to avoid carnivore attacks in the park. They conducted participatory methodologies with local communities afterward, noting that the community's approval [through a participatory approach] is key to the sustainability of any mitigation initiative (Kolinski and Milich, 2021). Although these studies' highlights are site-specific, the implications of different participatory approaches seem to help advances in mitigation initiatives more generally.

Globally, there is a growing research body addressing conservation conflicts and coexistence. Many studies point to the stakeholder's involvement as critical for a coexistence initiative's success (or failure) (Marchini and Crawshaw, 2015; Rodríguez et al., 2019; Badola et al., 2021; Cappa et al., 2021; Naha et al., 2021; Vargas et al., 2021; Yitayih et al., 2021). In fact, participatory approaches have not been used yet in human-wildlife contexts (Senbeto Jiren et al., 2021). Therefore, it is necessary to consider a multidimensional-systemic (social, economical, productive, ecological) or holistic approach to a conflict (König et al., 2020), especially in harsh environments such as mountainous areas with multi-species competitions and a rapidly changing global climate scenario. This holistic approach should conduct transdisciplinary (Reyers et al., 2010; Margules et al., 2020) and participatory work with stakeholders by co-producing territory knowledge (Von Wehrden et al., 2018; IUCN, 2020a; König et al., 2020; Senbeto Jiren et al., 2021). Conservation conflicts reflect poor land planning and high vulnerability to sudden environmental changes or extreme climatic events. However, conflicts could lead to dialogue opportunities on how stakeholders imagine their territory, manage it, and coexist with wildlife.

The Central Andes of Chile is part of the Mediterranean ecosystems, a dryland representing <5% of the Earth's surface (Hernández et al., 2015). These environments present high endemism across all spatial scales, identified as biodiversity hotspots: areas where human impact threatens many endemic taxa (Cowling et al., 1996; Myers et al., 2000, cited by Cowling et al., 2014). One of the most significant economic activities in the Andes mountain range is the adaptation of the ancient transhumant pastoralism practiced by local indigenous people (Marchant, 2019; Razeto et al., 2019). In pre-Columbian times indigenous pastoralists performed camelid movements through the Andes toward high-mountain grasslands. Afterward, Hispanic colonists replace domestic camelids with cattle, sheep, and goats (Razeto et al., 2007; Dong et al., 2016). Pastoralists move to livestock to graze in large-scale systems in this harsh environment, and through different sectors, with poor grazing planning and management. Over the years, pastoralists lowered their livestock economic dependence by finding secondary rural activities. Thus, they evolved into a new concept: agropastoralism (Dong et al., 2016). The agropastoralists maintain the seasonal movement of their animals (transhumance): keep the livestock in the valley during the winter and spring, and when summer comes, they take the animals to the highlands, where moisture is still optimal for fodder growth. However, the Mega-drought has impacted all the central areas of Chile since 2010, including the mountainous regions (Garreaud et al., 2017; Boisier et al., 2018), affecting transhumance. The concept of Mega-drought (defined by Garreaud et al., 2017) in central Chile refers to dry years' uninterrupted period between 2010 and 2015 (CR2, 2015; Garreaud et al., 2017). The Mega-drought began in 2010, with an annual rainfall deficit ranging between 55 and 75% in central Chile (30–38 S) lowlands, the contiguous Andes cordillera, and even westernmost Argentina (Garreaud et al., 2017). New studies point to the emergence of an upper-ocean warming area (termed the Southern Blob) as a significant contributor to the Mega-drought) (Garreaud et al., 2021). Several projections show a decrease in runoff due to climate change in central and southern Chile (Vicuña et al., 2010; Bambach et al., 2019). Therefore, mountainous areas' productivity has decreased, leading to significant animal (livestock) losses (López, 2019) and increased conflict between agropastoralists and wildlife. Many agropastoralists are abandoning their traditional livelihoods.

Pursuing coexistence became an urgent issue in Central Andes' mountain ecosystem, where growing human-wildlife conflict situations and environmental changes occur. Nevertheless, there is little scientific information about evaluating the baseline scenario of multi-species conflicts imbued in remote and large extension areas. For this purpose, participatory mapping could help to co-produce traditional ecological knowledge. Participatory mapping is a tool for gathering information regarding natural resources and local perceptions within a shared territory (Puri, 2011) and allows stakeholders to convey the location of activities and conflicts within the land (Basupi et al., 2017). It will enable both researchers and community members to examine the different views and uses of the socio-ecological system. The information analysis can allow science-based and socially sound land-use decisions (Raymond et al., 2020). It is predictive of both land use conflict and resolution (Brown et al., 2016).

All those above led us to co-produce territorial information with a transdisciplinary and participatory approach to orient decision-making toward a coexistence scenario between agropastoralists and wildlife in the Aconcagua Valley in Chile's Andes mountains. We conducted a participatory mapping methodology with a representative group of agropastoralists from the Aconcagua Valley (Andes mountains of Chile). We assessed their territory perceptions to address the baseline scenario of these particular multi-species conflicts. These cases are representatives from agropastoralists or farmers from the north-central region of Chile. This scheme sets the urgency for developing a participatory strategy to build territory planning, acknowledging the importance of local voices. Additionally, we explore how diverging arguments of stakeholders could orient decision-making processes toward coexistence in multi-species ecosystems.

MATERIALS AND METHODS

Research Site and Target Group

The research sites are located in Putaendo and Piguchen villages (**Figure 1**), within the upper section of the Aconcagua Valley (Valparaíso Region, Chile). The Aconcagua Valley's weather could be divided into High mountains' cold weather (Andes Mountain) and valley warmer weather (Cerda, 2016). The Aconcagua Valley has significant mountain heights that exceed 5,000 masl (Cerda, 2016).

We reached two agropastoralists' associations of the Aconcagua Valley, one from the Piguchen village (we will refer to them as the Piguchén association) (−70°41′O −32° 34′ S) and the other from Putaendo (we will refer to them as the Rinconada de Silva association) (−70° 71′O −32° 62′ S). There were 51 members (90% men) in the Piguchen association and 43 (100% men) in the Putaendo Association in 2020. Both associations work with livestock, and their primary pasturing system is seasonal exploitation of the grasslands: transhumance. The agropastoralists divided the grazing areas into two: (**Figure 1**). (1) Summerland ("veranada"): high mountain pasture areas used in the austral summertime (December to April), when moisture is optimal for forage growth. Summerlands are primarily managed and used by privates (i.e., local communities and mining companies). These grazing areas exhibit unique biodiversity value and are of particular concern to conservation biologists and conservation institutions. Summerlands also provide habitat and connectivity to relevant fauna species, including apex predators, puma (*Puma concolor*), and wild herbivores, such as guanaco (*Lama guanicoe*) (**Figure 3**). Their primary fertile spots are the "vegas," a type of high-mountain wetlands commonly found in the central Andes. They exhibit forage production, nourished by underground streams. They are strongholds of unique biodiversity, and they play a critical part in transhumance (Squeo et al., 2006). (2) Winterland ("invernada") (May to December): a lowland grazing area for keeping the animals safe during austral wintertime. These sites are relatively well-connected to urban areas. Their vegetation is primarily

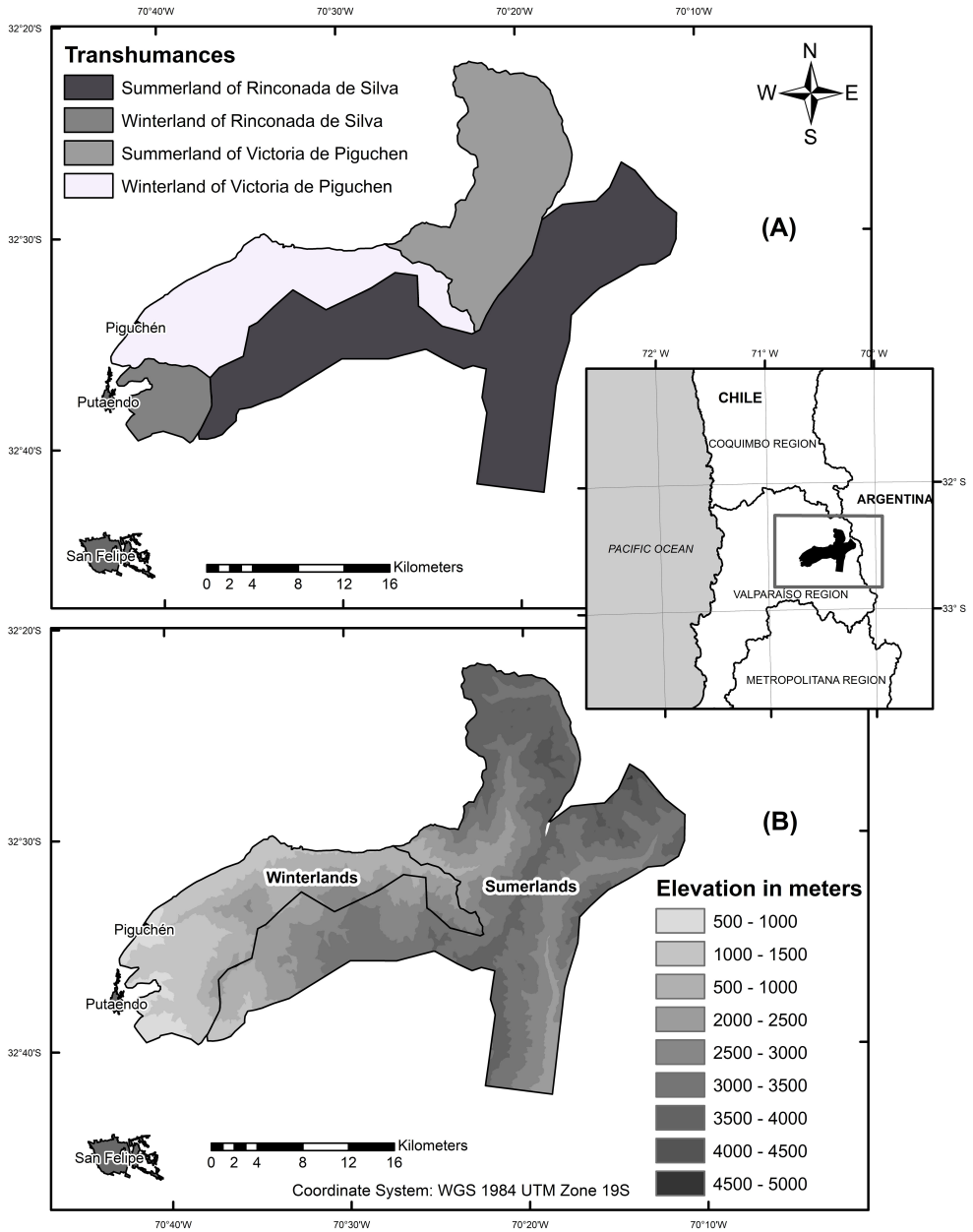


FIGURE 1 | The research site is located in Central Chile. **(A)** Represents the total area for transhumance activity. For each agropastoralist community. With their respective Summerland and Winterland. **(B)** Represents the elevation (MASL) of the research site for Summerland and Winterland.

bushes such as *Vachellia caven*, Mediterranean grasslands, and native trees. A lack of snowfall allows livestock to stay in the Winterlands over the autumn, winter, and part of the spring.

Agropastoralists keep the livestock in the Winterland from the beginning of autumn until late springtime (whereas Summerland is under extreme weather conditions). Then they move the animals back to the Summerlands, where temperatures and moistures spring belated forage production. This way, the transhumant system gets an extended grazing season for the livestock (**Figure 3**). In this activity, conflicts between agropastoralists and wildlife have intensified and become a significant threat in the last decade in central Chile (2010–2020). Possible causes are drastic climate change, the Mega-drought (explained above) (Vargas et al., 2021), urbanization, and land-use changes (Bonacic et al., 2007; Rodríguez et al., 2019).

Over the last years, urban and peri-urban inhabitants have increased the number of puma sighting reports to the public services (Bonacic et al., 2007; Sepúlveda et al., 2016). These agropastoralists link the puma-livestock problem with the hunter-prey interaction between these two groups. Nevertheless,

it has been described that free-ranging dogs are a significant threat to livestock production and wildlife (Muñoz and Muñoz-Santibañez, 2016). Most studies addressing human-wildlife conflicts refer to carnivores and livestock predation conflicts (Ohrens et al., 2016; Rodríguez et al., 2019). However, conflicts with the Andean condor (*Vultur gryphus*) (a large scavenger bird) and the guanaco (*Lama guanicoe*) (a wild herbivore) (Vargas et al., 2021) have drawn more attention recently. The Andean condor supposedly performs attacks on newborn calves during the birth season in the high mountain. Still, this fact would require more scientific observation to be confirmed. However, puma and condor impact on livestock production seems to go back several years. In both cases, agropastoralists blamed the wild species for the economic losses. These wild species are often hated and considered pests (retaliation measures have been documented, mainly for puma). The conflict describes an increase in these wild species' populations, thus competing with livestock, primarily cattle, for the available forage. Recent research shows how guanaco's conflict range is expanding through the whole Andes Mountain area of the Valparaiso region. Historically, this wild herbivore inhabits mountainous areas. Since pre-Columbian times, guanaco was used for its meat and leather. It also had cultural and mythological relevance for Andean Indigenous communities (Garrido, 2010).

The local institutions in charge of farmers' issues are the Agricultural and Livestock Development Institute of the Ministry of Agriculture (INDAP) and Servicio Agrícola y Ganadero (in livestock and agriculture, SAG). The latter is also the only institution that addresses conflicts with wildlife. The SAG receives wildlife-livestock attacks or predation reports and proceeds from identifying the wildlife species. However, many attacks occur in the high mountains, far away from urban areas, so the authorities often miss them.

Workshops Methodology

We conducted two workshops in January and October 2020 with two different agropastoralist associations: Rinconada de Silva (Putaendo) and Piguchen (**Figure 2**). The workshops were spaced seven months apart due to the SARS-CoV-2 (COVID-19) pandemics. First, we contacted the community leaders to explain the methodology and its objectives. They agreed that the method was in their best interest, so we decided to conduct one meeting with each community (to carry out the workshops). The communities are hierarchical, so any group calling for meetings must be extended through the leaders. In 3 years working on this site, we confirm that making a community invitation is valid and sufficient to achieve a significant quorum ($n = 84/94$), Rinconada de Silva ($n = 43$), and Piguchen ($n = 41$). The participants did not provide their ages.

We applied participatory community methodologies as tools for collecting information regarding natural resources and local perceptions within a shared territory (Newing et al., 2011) and attending to the land's local production and environmental aspects. In **Figure 2**, we explain the process for collecting information and analysis. Through participatory mapping in each workshop, we collected the following data: (1) mapping on natural resources or essential elements for the territory (raw map,

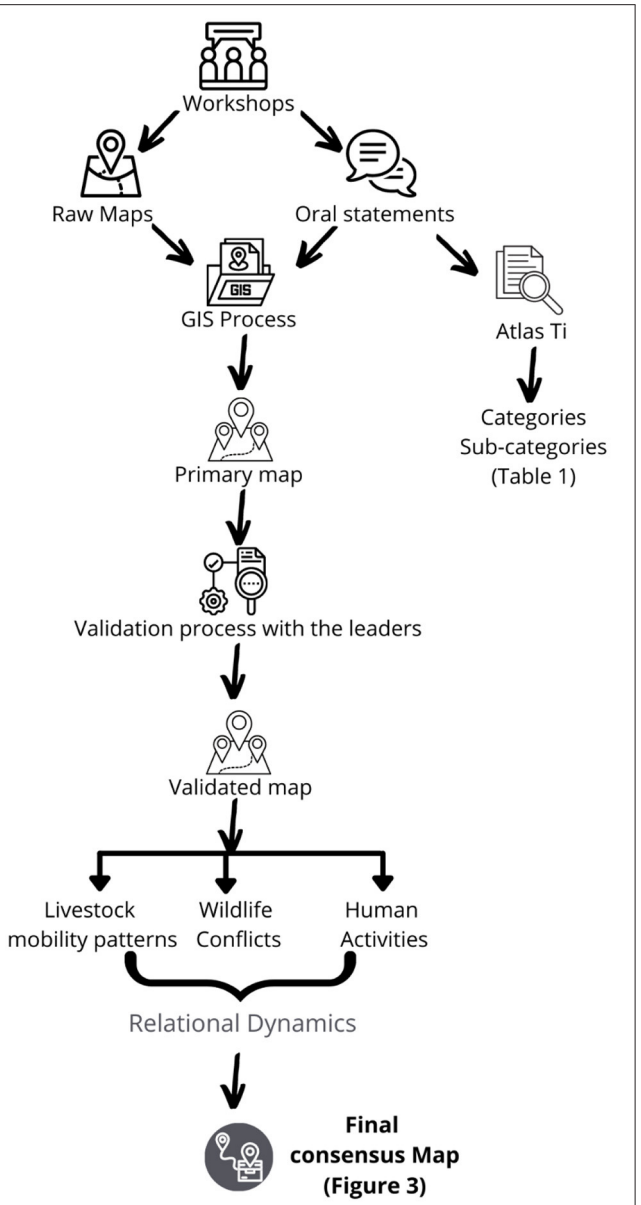


FIGURE 2 | Process diagram for obtaining a final consensus map from participatory mapping workshops.

in **Figure 2**): (2) evaluation matrix on the land and the natural resources use. In the evaluation matrix, we asked them to assign a score from 1 to 5 for temperatures and rainfall from the present time and 20 years ago. Finally, we asked them to declare "20 years from now scenario," exhibiting an "ideal future scenario": how much precipitation and temperature are optimal for their livestock production. We used the evaluation matrix to evaluate the perception of significant changes in environmental variables for their productive activity (Geilfus, 2002). The variables were: (1) temperatures, (2) rainfall/snowfall, (3) how these changes have affected the livestock activity in the last 20 years, and (4) how they relate these changes with wildlife. Agropastoralists mapped

these variables based on maximum and minimum thresholds for rainfall and temperatures. The maximum threshold was the most significant rainfall and temperature they remember. In contrast, the minimum threshold was the least rainfall and temperatures they could remember. With that settled, they draw many raindrops and "T" accordingly (**Table 2**). Agropastoralists explained the variable (3) and (4) in oral statements at the workshop (**Figure 2**).

These workshops aimed to broaden our knowledge of how human-wildlife conflicts originate. Participatory mapping assesses the community's Local Spatial Knowledge (LSK); it gathers spatial information to represent the reality of an organization or group of people (McCall, 2021). Participatory mapping of natural resources builds a visual representation of land use and natural resources perception. IFAD (2009) defined this methodology as information gathering made by the community in an open process, representing their real needs. This methodology attempts to gather information about a specific territory and its valorization by the community and then use it as a driver for decision-making and identify and locate the conflicts in such use.

The participants were randomized and divided into four groups of 4–7 people. We randomized the participants to deal with the communities hierarchy. By randomizing the participants, the leaders could fall into any group (we did not consider it socially appropriate to ask them not to participate). We did it this way to minimize the bias and have data representing reality beyond the leader's views (solely). Each group designated a group leader. We had a general facilitator and two supporting facilitators per workshop. We did this to ensure each group had a homogeneous dialogue, avoiding one-sided opinions and views (e.g., from the leaders). Each group was provided with materials and told to draw a map representing their territory as they perceived it, focusing on the winter and summer land use. We asked them to include three elements into the map: (1) natural and human components of the landscape (e.g., rivers, roads, forests, croplands), (2) areas with direct conflict with wildlife (puma, condor, and guanaco), and (3) areas with active use for grazing (Important for livestock activities). At the end of the given time, each group briefly exposed the results they obtained (oral statements). We documented the whole process by taking notes and photographing. At the end of each workshop, two sources of information were obtained that complemented each other: raw maps and oral statements. The oral statements provided complementary information to the mapping process: (1) attributes data, (2) non-spatial data, where otherwise [besides the oral statement] would not have been possible to have mapped. The low literacy level and the rural educational context make graphical representation challenging. Therefore, raw maps and oral statements are complementary and necessary to understand the productive system comprehensively.

Data Processing and Analysis

Consensus Map and Categorization of Attributes

We integrated the raw maps of each association into a single map. Each raw map contributed different attributes to the primary map (attributes described in **Table 1**). In two steps, we georeferenced raw maps data in the Geographic Information System (GIS) platform. First, the information of raw maps was georeferenced and identified by site names associated with two relevant spaces (Winterland and Summerland). Second, we used group information to complement a new map (primary map) (one for each association). Then we validated the primary map with the leaders, obtaining a validated map. Then, the attributes were classified into three categories to get an accurate visual representation: (1) Human activities; (2) Wildlife presence and interactions sites (human-wildlife conflicts); (3) Livestock mobility patterns within agropastoralists territory. These attributes represent elements in constant interaction within a mountainous system. The outcome of the process diagram (**Figure 2**) is a final consensus map representing the perception of these attributes' relational dynamics. With this information built, we elaborated the map presented in **Figure 3**.

Data Analysis

The notes of oral statements in Atlas.ti (2018) identify participants' explanations of the territory's use and threats, using Grounded Theory procedures (Strauss and Corbin, 1990). First, we reviewed the notes taken during each map explanation and compared them with the map's spatial information. After that first stage, we generated a set of categories we thought could group all responses. After discussing and editing the categories so they wouldn't overlap, we had a definitive set of six (6) categories and every response belonging to one of them. After that was accomplished, these six categories were grouped into two types: (1) Threats to livestock activity and (2) Threat resolution pathways. We chose this method due to the heterogeneity of the information obtained from the workshop. We systematized it to make it helpful in understanding Human-Wildlife Conflict. We identified the critical graphic features to put in the territory represented by icons, based on Burkhard and Maes (2017) recommendation for analyzing cartography. We conducted this process with each community and synthesized it as one map for the agropastoralist association.

RESULTS

Results show the participation of 84 members from both associations, 43 from Rinconada de Silva (100% men) and 41 from Piguchen (97.5% men). Agropastoralists identified four key sub-categories for the 'threats to livestock activity and two categories as "threats resolution pathways" (**Table 1**). The following sections describe agropastoralist perceptions of each sub-category. Some of the raw maps are available in **Supplementary Material**.

Threats to Livestock Activity

Climatic Conditions

According to the agropastoralists, the shifting climatic conditions were the most relevant threat to livestock activity (**Table 1**). It directly affected livestock production because of the loss of primary productivity in grazing areas and water springs turning into swamps.

TABLE 1 | Categories and sub-categories emerged from the workshop's response analysis.

Category	Sub-category	Spatial attribute	Attributes name	Description
Threats to livestock activity	Climatic conditions	Yes	Temperature and snow	Mainly drought, expressed in a decrease in rainfall and snow and primary productivity of grasslands
	Interactions with wildlife	Yes	Fox, puma, guanaco, vulture	Predation (puma, fox, and Andean condor), and resource competition (Guanaco)
	Anthropogenic threats	Yes	Mining, dogs	Human activities perceived as threats: cattle theft, large-scale mining, motorcycling in grazing areas, free-ranging dogs
	Relationship with the State	No		Refers to the interaction between agropastoralist and different state agencies: technical assistance programs, SAG and police force, and municipality
Threats resolution pathways	Adaptation of agropastoralist's activity	No		Refers to new practices, or changes in current practices that would allow them to continue raising cattle in those lands
	Complementing agropastoralist activity	Yes	Recreational tourism	Activities that could be developed within the land destined for grazing but that are not livestock activities

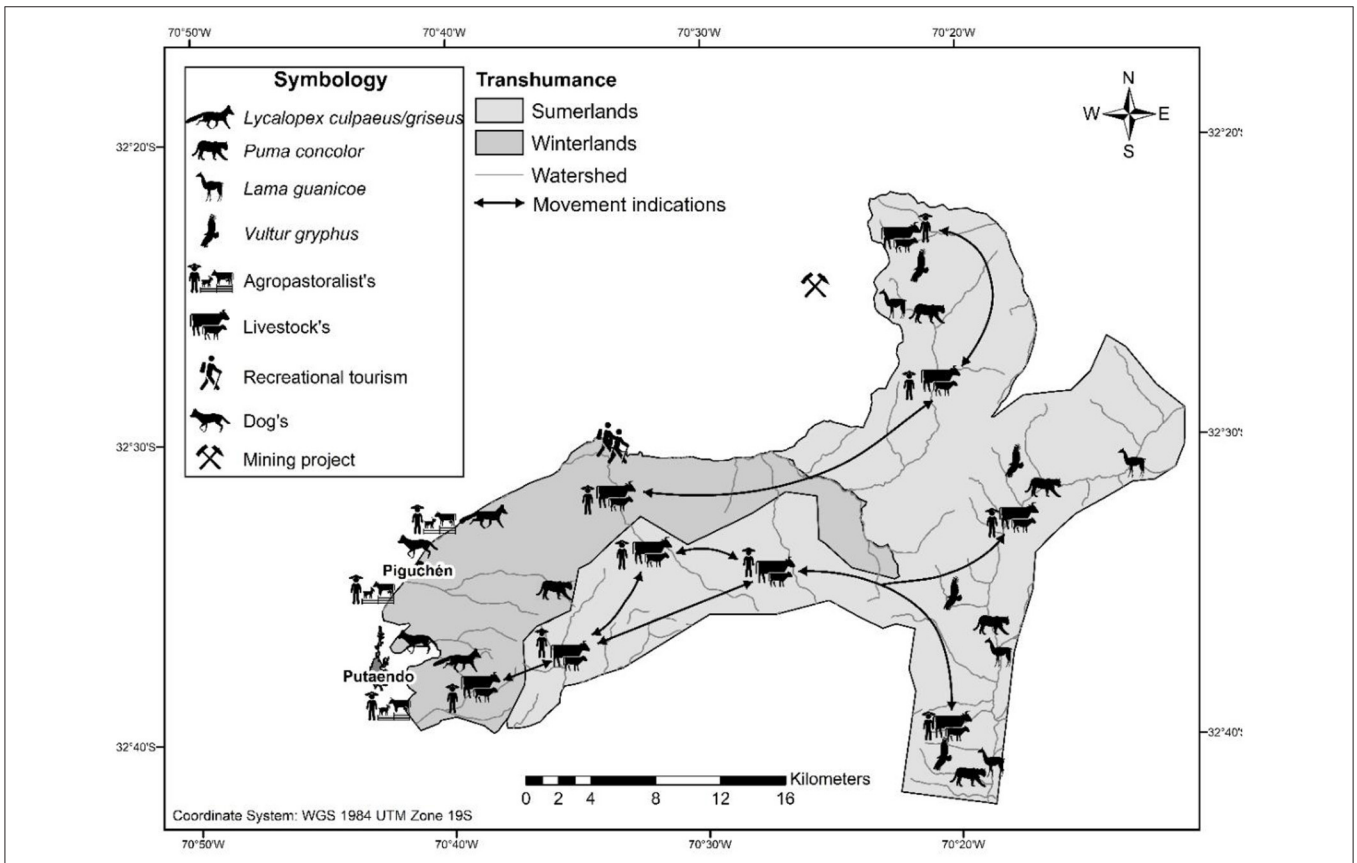


FIGURE 3 | Final consensus map. It indicates the mobility patterns of the livestock and agropastoralists between Winterlands and Summerlands, localization of wildlife, and other human activities such as recreational tourism. In the Winterland, livestock interacts with wildlife such as *Lycalopex culpaeus/griseus* and human activities such as recreational tourism. In the Summerland, livestock interacts with wildlife such as the *Puma concolor*, *Lama guanicoe*, and *Vulture gryphus*.

Both associations gave their perception of the rainfall and temperature shift from the past (20 years) to the present (2020) (**Table 2**). Drops and "T" represent Rainfall and Temperature, respectively, and the differences are proportional. The agropastoralists noted their perceptions within two thresholds: maximum rainfall ever recorded (the most significant

TABLE 2 | Evaluation matrix.

Association	Group	Rainfall		Average temperature	
		Past	Present	Past	Present
Rinconada de Silva	1	●●●●	●	T°T°T°	T°T°T°T°T°
	2	●●●●●	½●	T°T°T°	T°T°T°T°T°
	3	●●●●●	½●	T°T°T°	T°T°T°T°T°
	4	●●●●●●	●●	T°T°	T°T°T°T°
Piguchén	1	●●●●●	●●●●	T°T°T°	T°T°T°T°
	2	●●●●	●●●	T°	T°T°
	3	●●●●	●●●	T°T°T°	T°T°T°T°T°
	4	●●●●●	●●●●	T°T°T°	T°T°T°T°

Quantification of the perception of significant change in environmental variables (rainfall and temperature) regarding past (20 years) and present conditions (2020).

number of drops in the table) and minimum rainfall ever recorded (the least number of drops), same for the temperature.

All the groups agreed that rainfall significantly decreased in the past 20 years in both associations. Groups 2 and 3 from the Putaendo association declared a 90% rainfall decrease. Data for temperature shows that 100% of the associations' groups perceived that average temperatures have increased over the last decade. They marked the Summerland as the place where changes are most observable. They perceived temperature changes through two indicators: (1) A greater thermal amplitude between night and day, and (2) A reduction in the snowfall period and an apparent acceleration in the melting of the snowpack. In the spoken testimonies, agropastoralists from both associations declared hope for an increase in rainfall (ideal future scenario). Still, they think it will decrease in the next 20 years (future scenario). In the average temperature perceptions, each association group expects an increase in future average temperatures. Each group from both associations shows concern about climate change's adverse effects. Both associations said that Mega-drought is the primary current factor threatening their productive system.

Every year agropastoralists decide whether to take the animals toward the mountains. This decision depends on a previous evaluation of high grasslands. The evaluation criteria may vary depending on several factors, such as the geographical area, vegetation productivity, and personal considerations from the agropastoralists. They declared that it is unlikely to take the animals to the Summerland this year (2020) because the "risk is too high".

Interactions With Wildlife

Both association groups pointed to the interactions with wildlife as a significant threat derived from the use of Summerlands. (1) Predation: they accuse predator attacks from puma to every livestock species they own (cows, sheep, goats, horses, and mules). Agropastoralists also declared that Andean condor performs attacks on cattle, especially on calves. As the agropastoralists described, Andean condor attacks occur when an individual or a group of condors separate and haunt a calf toward a cliff, causing it to fall and die, to then scavenge the remains. (2) Grass competition: the associations also declared an existing conflict for grass with the guanaco, especially in high-mountain areas (vegas). Agropastoralists said that the guanaco consumes "their property" (the grass) in the Summerland before they get to reach the same place with their animals. This situation would be affecting their animals' nutritional health. Both types of conflict, predation and grass competition, could be labeling the Summerlands as the most conflicting area. They also reported some minor damage performed by foxes, affecting only sheep and goats in Winterland. Parallelly, they have admitted that wildlife is a part of the mountain ecosystem, and they need to learn to live with it.

The association groups emphasize climate change and increased causality of wildlife conflict (especially with guanaco). Both groups declare that the decreasing of the snowpack clearly benefits the guanaco. They explain that high-mountain snowfall used to be more aggressive in previous years (10 years), thus controlling guanaco births, especially in wintertime. At the end of winter, the agropastoralists used to find frozen guanaco bodies in the high mountain. Recently they have not found any. This situation could reflect a climatic event promoting the guanaco's survival and possibly affecting its population. Puma and condor seem to have a more remote interaction with the agropastoralists. The latter explained that they perceive a stabilization in attacks performed by these two species. However, the livestock flow to the Summerland has decreased, increasing the proportional damage per predator attack. Every animal loss has a more significant impact on flock size.

Anthropogenic Threats

They also reported Mega-mining projects prospecting near the Summerland valleys as a threat to agropastoralists. Mega-mining, they said, represents a significant loss of space for pasture and disturbance of the natural environment. Another anthropic threat in both Summerlands and Winterlands is cattle raiding and free-roaming dogs. In addition, there have been reports of illegal drug activity in the Winterland area, which is declared as an emergent threat (currently not causing conflict). Finally, in the Winterland, unregulated activity such as motorbiking is reported as a severe threat to foraging availability since it erodes the already fragile hillsides and gullies. Regarding these threats, both associations are worried, but only the Piguchen association signaled to be directly affected by it.

Relationship With the State

The agropastoralists described the relationship with state institutions as a problem. The agropastoralists perceive treatment from the SAG and the police as unfair. They say that these institutions do not take sufficient actions when cattle raids or depredation (by puma) occur in the Summerlands. On the contrary, the associations declared that SAG often blames them for guanaco poaching accusations. Agropastoralists also described the technical assistance from public agencies as insufficient and irrelevant. These factors led to a feeling of abandonment from the state.

Threats Resolution Pathways

Adaptation

In the threat resolution pathways, the most relevant category was "measures to adapt their activity to the changing climatic scenario." The most pertinent adaptation idea was changing the grazing management in Winterlands toward a more resilient one. Then, the resource can last longer and reduce the dependence on Summerlands. The associations stated that Summerland's grass production is progressively becoming less reliable, so they have to analyze every season whether or not to include it in the grazing program. Both associations declared their availability to collaborate in searching for productive alternatives to Winterland.

Supplementary Agropastoralist Activities

Results show that the Piguchen association would consider doing additional activities such as mountain and archaeological tourism as a possible way to diminish the threats they face. The particular characteristics of the territory make it potentially attractive for tourists. Piguchen association considers ecotourism in the high Andes as a possibility, particularly in the Summerland area. Two critical reasons reinforce this argument: (1) Ecotourism will complement the high Andes' agropastoralism (Only if the environmental conditions are stable) (2) Re-value the territory and the natural heritage.

The participatory mapping results reflect different perceptions among the participants on the importance of the territory, such as the natural borders in each grazing sector, access to specific sites, and the mining companies' relevance. However, 100% of the groups agreed that the Mega-drought was the primary driver threatening their production, increasing the conflict with wildlife. Thus, some groups declared different aspects for the "ideal future scenario" for the Aconcagua Valley. All the participants said, "Any future will only be possible if more rain (or water) comes." Finally, they have considered the Summerland as an area of high risk due to multiple factors such as the conflict with various wild species. Considering the climate change scenario, they have decided to decrease the use of Summerland and improve the Winterland management planning to increase livestock productivity and reduce conflict with wildlife.

With the attributes presented in **Figure 3**, we built a summarized representation (**Figure 4**). We showed how the agropastoralists perceive the connection between wildlife interactions, significant changes in climatic variables, and other anthropogenic land uses. We stress the general perception that the livestock rearing activity is currently affected by external factors (e.g., climate, grassland production, interaction with wildlife). The current Mega-drought scenario led to a perception that the traditional agropastoralist activity is in crisis and needs external aid.

DISCUSSION

In this study, we identified prioritized areas to mitigate conflict with wildlife. To achieve this goal, we build a participatory mapping with local communities. Summerlands areas were identified as critical areas to reduce human-wildlife conflicts in the study area.

Our study reveals the local livestock-wildlife conflict as one part of a much more significant climate crisis. Our study provides information about current challenges faced by transhumant livestock activity and its impact on the interaction with wildlife. We summarize the main aspects of agropastoralist perceptions about territory, interactions with wildlife, and climate changes associated with the Mega-drought scenario (**Figure 4**).

Climatic Conditions

Our findings showed that agropastoralists have already perceived climatic changes in the mountains of Aconcagua Valley. Consistent with Roco et al. (2015), agropastoralist recognize a change in temperature and precipitation in the recent past. However, our study it's the first to show perceptions of climate changes in the mountains of Central Chile, while Roco et al. (2015) showed results with Family farm agriculture (FFA) in valleys of central Chile. These kinds of agriculture don't use the high mountains for pasture. Studies from other mountainous areas show similar findings (Batumike et al., 2021) regarding increasing temperature and decreasing precipitation. These results are essential because scarce information has documented pastoralist perceptions about climates changes in mountainous areas, most of the studies focused on agricultural communities (Reyes-García et al., 2019; Batumike et al., 2021).

Table 2 also shows variation in perceptions among associations. This could be because of differences in the age composition of each group, resulting in different amounts of time spent experiencing the rangelands and, therefore, different perceptions of climatic variation, but further research would be needed to confirm this hypothesis. The variation in the final consensus map would not be expected since it compiles every element identified by each group within their shared territory.

Our results show that the ideal climatic scenario for both associations (100% of the participants) leads to more precipitation and a lower average temperature. However, they declare to be aware of how unlikely that future is. They think that rainfall will decrease and the temperatures will increase. In fact, climatic projections for the area are not promising. Aldunce et al. (2017) show that annual precipitation is unlikely to grow, and the Mega-drought is likely to continue. Hundred percent of the participants claim that they will continue with the transhumant activity for several reasons: tradition, economy, emotional attachment, reasons previously described by Razeto et al. (2007). These motivations could reflect a relationship between the agropastoralists, the transhumant activity, and the mountain environment, beyond merely the economic.

Like our study, in the north of the studied area, farmers' beliefs suggest that the Mega-drought and climate change are underlying factors contributing to the farmers-guanaco conflict (Vargas et al., 2021). The changing environmental conditions seem to expose livestock activity to greater vulnerability, affecting wild species' tolerance. Regarding the climatic conditions, the projections coincide with the farmer's arguments. Central Chile has been facing a rainfall decline during the last decades. Persistent drought showed a precipitation deficit of

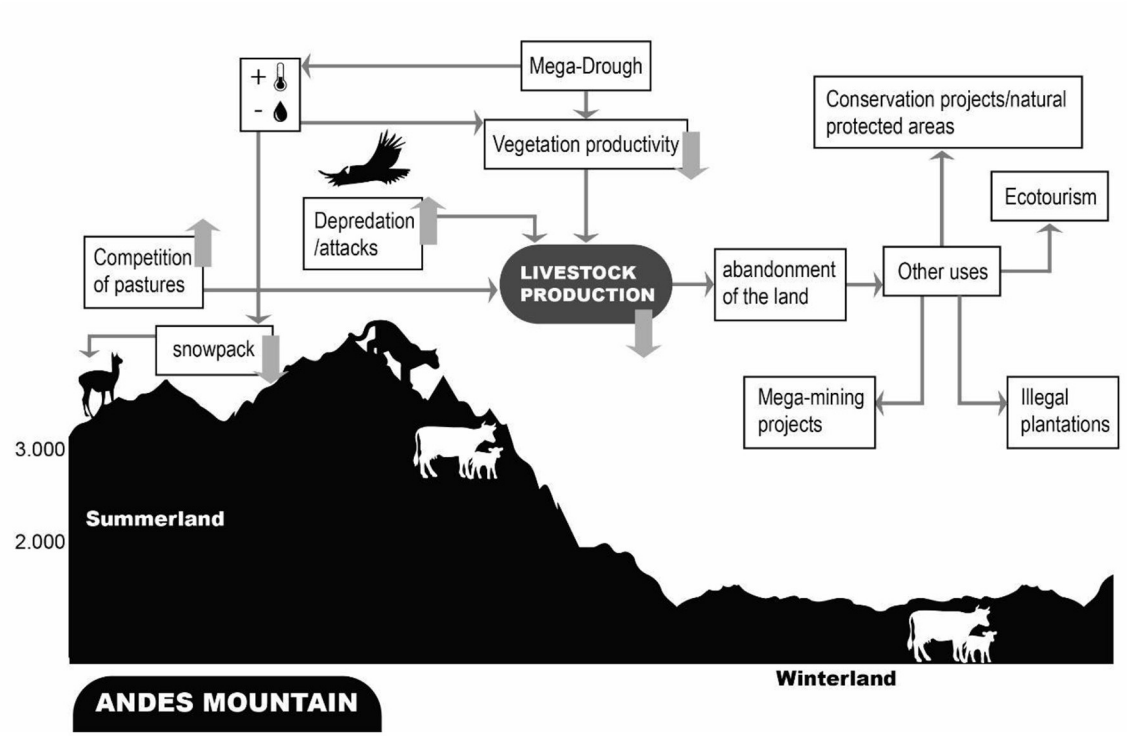


FIGURE 4 | Visual representation of agropastoralists' perception of their territory (drawn from the workshops), divided into Winterland and Summerland. The image reflects a complex scenario, with multiple potential interconnections among elements, ultimately affecting livestock production: (1) Interactions between livestock and wildlife in the Summerland result in competition for pastures or depredation and attacks, depleting livestock production. (2) High temperatures and less rainfall led to the snowpack's thawing and decreased vegetation productivity, potentially intensifying competition for pastures. (3) Less livestock production results in more agropastoralists leaving the territory, thus allowing other activities to take over, seizing the unattended land.

∼30% (Boisier et al., 2016). Recent studies highlight that drying and warming conditions are very likely to continue with substantial impacts on surface hydrology, vegetation productivity, and snowpack (projected to decrease by 35–45%) (Bozkurt et al., 2018). The projected changes could significantly impact local agropastoralists' socio-economic conditions and wildlife interactions. A recent study reveals that the current Mega-drought is the hardest in the last 600 years in Central Chile (Morales et al., 2020). It suggests that South America will experience more severe droughts (IPBES, 2019). Despite this, no studies reveal the impact of climate conditions changes in Central Chile's mountain biodiversity or the consequences (direct and indirect) in wildlife interactions with other activities.

Interaction With Wildlife

Our results show that wildlife affects agropastoralists throughout the year but in different areas. Most of the reported conflicts regard carnivores and scavengers. However, there was an increasing conflict with guanaco in the last years. Perceived triggers of conflicts with each species are different. The agropastoralists perceived an increase in the guanaco population, possibly leading to direct competition for fodder with the livestock.

Regarding the puma, the problem seems connected to another issue besides predation. There are fewer agropastoralists present in the mountain because of the drought. So, one single attack performed by a puma could seem proportionally more damaging (more puma livestock ratio).

Nevertheless, there is little scientific information on guanaco's population estimates: the current data is scarce and outdated, difficulting a decision-making process. Regarding carnivores and scavengers, agropastoralists also declare that although the puma's population has not risen, the attacks performed on livestock (especially small ruminants) have increased. Currently, some agropastoralists are no longer willing to use the Summerland because the economic damage is "too high" (near 70% of losses in some cases). Despite the yield consequences, they would prefer to keep the flock in the Winterland if they have mainly small-ruminants. We encourage more and new research that goes deeper into this matter. Future studies should address wildlife population shifts and their interaction with agropastoralists.

Based on agropastoralists' experiences, we identified Summerlands as a high-risk area for their practices. The primary drivers to this risk could be the problematic interactions with wildlife, other human activities, and the declining vegetation productivity due to the Mega-drought. The associations marked the Mega-drought as the primary driver of this degradation process. This reality seems compatible with recent findings that climate change intensifies human-wildlife conflict by exacerbating resource scarcity (Abrahms, 2021). Additionally,

climate change reinforces the Mega-Drought (Garreaud et al., 2021). Based on agropastoralists' perceptions, our results follow other scientific results that the frequency and severity of human-wildlife conflict are rising, especially in Summerlands areas or more degraded areas (Nyhus, 2016; Hodgson et al., 2020).

Anthropogenic Threats

Besides the environmental changes, anthropogenic actions could harm transhumant agropastoralism. And these changes could affect the interaction with the wildlife species.

Recently in the valley, there was registered some external activity: (1) Mega-Mining projects at starting operations, (2) Cattle raiding and free-roaming dogs, (3) Reports of illegal drug activity in the mountain (near to Winterland area), and (4) reported recreational motorbiking activity. It would be necessary to monitor how these events unfold over time and interact with agropastoral activity and wildlife. Based on other studies, we tend to project that these activities could impact transhumant pastoralism and the environment as a whole.

Mega-mining projects and other extractive activities adversely affect ecosystems by destroying habitat, overexploiting, polluting, and creating species disbalance and disease (Carranza et al., 2020). The level of production defines Mega-mining projects. This concept particularly references a large-scale extractivist output carried out by multinational companies, which occupy large extensions of territory for their development (SONAMI, 2014; OLCA and CGCGA, 2016). Mega-mining projects effects directly impact the transhumant agropastoralist activity.

As we have seen in previous sections, transhumant agropastoralism relies on natural seasonal pastures. The mega-mining procedures would imply disturbing noises, the creation of bare soil, and pollution of the water sources (Mensah et al., 2015). Some of these impacts could be irreversible. The destruction of aggregated soil (already weakened by climatic conditions) would imply a significant shift in soil structure, depleting grass production and increasing erosion. These changes will also negatively affect the ecosystemic processes such as the water cycle, the mineral cycle, the community dynamics, and the energy flow (Savory and Butterfield, 2017).

Another threat to agropastoralists is cattle raiding and the presence of free-roaming dogs. Free-roaming dogs perform attacks on livestock, potentially causing significant animal losses, spreading diseases (zoonotic and interspecies), causing relevant disturbances to wildlife, and competing with endemic species (Young et al., 2011). It is critical to conduct further collective methodologies, with a participatory approach to put all stakeholders in the common ground to take action and prevent this particular threat from growing.

The presence of illegal drug activity in the mountain area (Winterland) represents a significant future threat. Illicit markets are complex, but drug trafficking is often linked to other activities, such as wildlife trafficking (Van Uhm et al., 2021). The Aconcagua Valley would be an ideal scenario. The mere presence of illegal drug activity suggests that a further abandonment of the mountain by agropastoralists could leave a 'free space' for conducting illicit activities.

Recreational motorcycle activity looms as a significant threat, with scalable negative consequences. The repeated wheel traction causes significant erosion in already fragile land (causing productivity loss). Vehicle-wildlife interactions are proven to generate Roadkill, a term given to a dead animal struck by a vehicle. This phenomenon could significantly affect wildlife populations (Chen et al., 2021). We suggest collective action to enhance local education.

Relationship With the State

The agropastoralists declare a complex relationship with State institutions, pointing them as irrelevant when dealing with the livestock-wildlife conflict. State institutions hypothetically could argue that the livestock-wildlife conflict is an emerging issue, so classified as an unusual activity (out of their routine duties). However, the agropastoralists noted that even for the routine duties, state institutions are ineffective.

The involvement of stakeholders (including the State institutions) is vital for a conflict to be correctly addressed or for a participatory approach to be successful (Hargreaves et al., 2021; Spratt et al., 2021). In this regard, it would be necessary to fully address the agropastoralist-state relationship issues. The state and agropastoralists have an unequal power relation, and the latter feel abandoned by the state. They claim the state should prioritize the conflict and offer some "resolution paths." The alleged lack of awareness by the state could lead to several scenarios: (1) intensifying the conflict, (2) retaliation actions. Collaboration between agropastoralists and other stakeholders (e.g., other communities, NGOs) could strengthen communication channels with the State (Coria and Calfucura, 2012).

Considering that the agropastoralists' perception exhibits only one side of the conflict, it is imperative to address a broader perspective with all the stakeholders to broaden our knowledge of the community's willingness to achieve a coexistence scenario (Akpo et al., 2015). A complete view will also accelerate the adoption of future territory planning (Spratt et al., 2021).

Adaptation

With our results, we suggest that the Winterlands' adaptation is the main pathway to address the threats agropastoralists face in their activity. This could be done through changes in current practices or including new methods in their activity to continue raising cattle in those lands. Since the association agreed on looking for alternative management, few workshops have been conducted to date (with few people due to the COVID-19 pandemic). However, these meetings have presented promising preliminary results. Due to the current state of degradation of the Winterland, we have concluded that adaptation to climate change should start by restoring this agroecosystem, at least partially (Del Valle et al., 2021). It is imperativeto initiate a participatory ecological restoration program in the Winterlands, so the recent livestock management shifts into one that will enhance the Winterland's resilience, improve productivity, and build adaptation to droughts. To do so, it is necessary to take elements from diverse disciplines, such as Ecological Grazing Management, Agroecology, and Ecological Restoration (Guzmán Luna et al., 2019). An interdisciplinary work can diminish grazing

pressure over the Summerland, which has become a place of high vulnerability due to the uncertainty about climate and forage (a consequence of climate shifts or competition for forage). This has exposed agropastoralists to significant animal losses (predation, abrupt changes in the weather, raids).

Agropastoralists are currently evaluating whether to use Summerlands or not, ever again. This scenario sets an opportunity for the conservation of wildlife in these spaces. However, this opportunity must consider restoring and adapting the livestock production to less susceptible areas, such as Winterland. The future of large native species such as the condor, the puma, or the guanaco (which require large amounts of habitat), depends on the ability to conceive a well-planned human-wildlife program.

Contrastingly, the primary adaptation-to-climate-change actions proposed by the government focus on facilitating access to credit for private investment for adaptation. In the specific case of dairy goats, advance toward complete stabling, subsidizing feed purchasing. There is no other action exact for livestock production in the National nor Agricultural Climate Change Action Plan (Ministerio de Agricultura, 2013; Ministerio del Medio Ambiente, 2017). Therefore, these actions do not match the local proposals collected in the workshops. Agropastoralists have not been consulted in any previous process of public policymaking.

Nevertheless, this neglect is widely known. As FAO states in its most recent study about agropastoralism strategies, governments often lack knowledge of integrated landscape management (Wane et al., 2020). Also, evidence suggests that every successful government policy enhancing local adaptation must address the agropastoralists' heterogeneity (Cuni-Sanchez et al., 2018). Decisions should emerge from positive linkages between local communities' knowledge and organization and higher-level institutions (Fu et al., 2012).

Supplementary Agropastoralist Activities

Results show that agropastoralists have significant knowledge of the mountain and its biodiversity. Thus, Razeto et al. (2019) define the agropastoralists (or Arrieros) as guardians of the territory. "As they live in it, they will take care of it." They also show that the Piguchen association would consider doing other mountain and archaeological tourism activities. This association declares that ecotourism could be a "resolution path," positively affecting family economy and tradition. Furthermore, it would allow the agropastoralists to improve their financial status while maintaining their way of life). Although it does not address the effects of the Mega-drought and the derived conflicts, ecotourism looms as a feasible possibility to face the financial urgency.

Lee and Jan (2019) define ecotourism as traveling to less exploited natural places to see the natural setting and wildlife and enjoy culture while conserving the site's environment. Some elements of that definition could be acquired for further discussion. On the one hand, ecotourism could help agropastoralists adequately protect the land (by providing financial revenue and increasing their presence in the mountain). Also, this activity could provide environmental education for a broader audience. In contrast, some risks in this activity could be public exposure, which could bring negative externalities, such as contamination and over-visit of fragile, sensitive areas (Collins, 1999, quoted by Khanra et al., 2021). Additionally, a complete turn from agropastoralism to tourism as their primary economic activity would make them dependent on tourism market oscillations. Suppose the tourism market drops as it did with the SarsCoV-19 (COVID-19) pandemics. In that case, it could constraint their livelihoods even more, so several parameters should be taken into account to determine whether or not it is profitable and ecologically relevant.

Results of the participatory mapping show that the agropastoralists own significant knowledge of the mountain (and its biodiversity). Thus, Razeto et al. (2019) define the agropastoralists (or Arrieros) as guardians of the territory. "As they live in it, they will take care of it".

Final Considerations

In summary, this study represents a contribution to human-wildlife studies because we put into value local knowledge and local perceptions to build territorial planning in the context of Human-Wildlife conflicts. We also value local perceptions about climate change impacts in the context of Megadrought (García-del-Amo et al., 2020; Batumike et al., 2021). Agropastoralism in Central Chile seems to be an activity where the agropastoralists do not constantly stay with the livestock. A lack of management can contribute to a greater risk of losing animals (raids, attacks by carnivores, etc.). Better management practices, planning, and livestock can reduce losses and increase agropastoralists' productivity. Our participatory mapping set up a reflection space for agropastoralists. They account for the current challenges of the activity and open space for coexistence. Participatory mapping can be understood as a "situation assessment," informing and guiding decision-making, particularly challenging study areas. This mapping allows us to prioritize places to start working and possibly take action. Notably, this case shows us how the Summerlands are highly challenging areas due to three main reasons: (1) mitigating conflicts with wildlife (2) they present uncontrolled anthropic intervention, such as livestock raids and mining (3) there are areas of significant climatic uncertainty. Results reflect an uncertain future for coexistence in this particular research site. The chronological analysis shows us how the agropastoralists projected their activity. And what the wildlife's role in the system is. Simultaneously, this methodology creates a space for ongoing reflection and debate on the activity's current productive and ecological issues. Agropastoralists' testimonies reflect that this reflection space is innovative for the area: they do not recall a similar previous process. Historically there have been few stakeholders, through non-socialized public policies, driving all the decision-making in agropastoralism at the research site. This situation could have led to a non-participatory approach to the site's issues so far, resulting in a persisting (and growing) human-wildlife conflict.

The current scenario shows a probable increase in livestock and wildlife interaction, a shifting climate, and an increase in agropastoralism vulnerability. Additional multisectoral actions will be required to prioritize the area's activity and adapt it to the future scenario. Future research is needed to increase resilience, reduce vulnerability and adapt the mountain agropastoralist

sector. This study also confirms that complex socio-economic, political, and environmental factors (and their interaction) drive negative human-wildlife reciprocity. Previous studies reveal the need to adopt holistic management for multi-species conservation conflicts (Pozo et al., 2020). The consequences have been (and will be) severe, especially in the agriculture sector and biodiversity. Territory threats represent a significant driver to the intensifying of the conflicts. Future initiatives in the area will only be possible if the state works with local communities and scientists to develop public policies that promote biodiversity rehabilitation.

DATA AVAILABILITY STATEMENT

The datasets presented in this article are not readily available because the studied group authorized us to publish only two raw maps (available in supplementary materials). We are not providing the complete set of raw maps in supplementary materials since they might have sensitive information regarding the community's privacy and the socio-environmental conflict with the mining company. Requests to access the datasets should be directed to SV, solangevargas@gmail.com.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

SV, MH, and JD: study design and data collection. SV, MH, JD, and AH: analysis. All authors: writings and revision. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fcosc.2022.731382/full#supplementary-material

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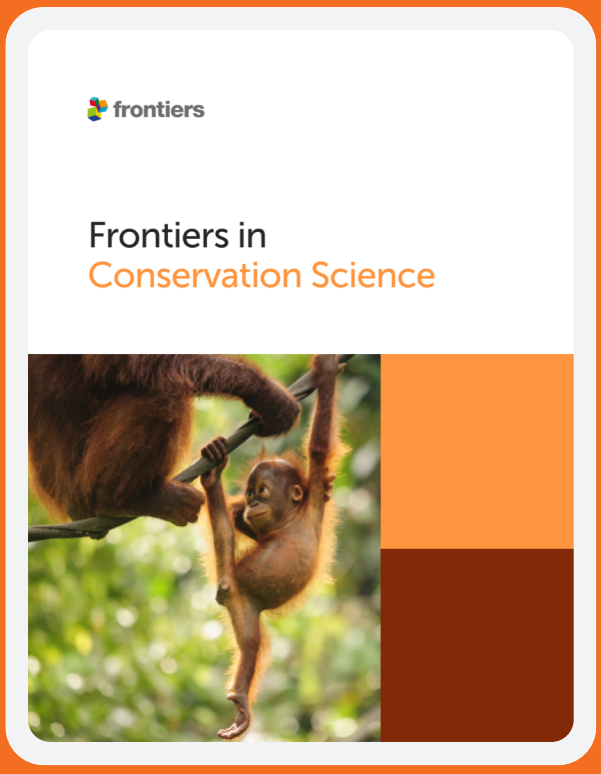
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